4th edition

NATURAL RESOURCE AND ENVIRONMENTAL ECONOMICS

ROGER PERMAN   YUE MA   MICHAEL COMMON   DAVID MADDISON   JAMES McGILVRAY

Natural Resource and Environmental Economics

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Natural Resource and Environmental Economics

Fourth Edition

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Contents

Preface to the Fourth Edition

As we wrote in a previous preface, there are two main reasons for producing a new edition of a textbook. First, the subject may have moved on – this has certainly been true in the area of natural resource and environmental economics. Second, experience in using the text may suggest areas for improvement. Both reasons warrant a fourth edition now.

We will say nothing here about the ways in which the subject area has 'moved on' except to note that it has and that you will find those changes reflected in this new edition. As far as experience in the use of this text is concerned, some more comment is warranted.

First, we have received a lot of feedback from users of the text. Much of this has been highly favourable. Indeed, the authors are very pleased to note that its readership has become very broad, a characteristic that has been enhanced by Chinese and Russian translations. User feedback – formal and informal – has provided us with many ideas for ways of making the text better. We are particularly grateful to those individuals who provided solicited reviews of the third edition, and to the many readers who made unsolicited comments. Many of the changes you will find here reflect that body of advice.

The invitation to prepare a fourth edition had an important bonus for the authors of the previous editions: they were able to ask Professor David Maddison to join the authoring team. David accepted the invitation, a decision with which the other authors were very pleased. In addition to adding freshness, additional insights and new perspectives to the book, David joining the authorship means that the average age of its authors has now fallen, which should provide greater security for the long-term future of the text. As environmental economists, sustainability is naturally high on our list of objectives.

The move from third to fourth edition has not altered the structure of this textbook in any significant way: in particular, we have largely retained the previous division into Parts and the clustering of themes. But the content has been substantially altered in several places. As with any new edition, the text incorporates substantial updating of material. This new edition also includes much additional content, both theoretical and empirical. And in line with the demands of readers and suggestions of referees there is a greater emphasis on environmental policy.

The chapter on environmental valuation has been substantially modified by the inclusion of new theoretical and analytical content. The discussion of international trade and the environment – which in the third edition occupied just one part of a chapter dealing with international environmental problems – is now given its own chapter, and covered in far greater depth as befits its importance in environmental and resource economics.

To accommodate the additional text that followed from those changes, we have taken the mathematical appendices out of the printed textbook itself and relocated them on the book's Companion Website. The collection of learning resources on that site is now far more extensive than in earlier editions, and we hope that this can facilitate the process of updating relevant material between future new editions. Further details of changes made in the fourth edition are given in the Introduction.

There are several friends and colleagues the authors would like to thank. We remain grateful to Jack Pezzey for writing an appendix to Chapter 19. Mick Common, Yue Ma, David Maddison and Roger Perman would like to express their gratitude to Alison McGilvray for her continued support and encouragement throughout this revision process. The genesis and early editions of the book owe much to her late husband, Jim. We hope that she would agree that this new edition is one of which Jim would feel proud.

David, Mick, Roger and Yue have succeeded in remaining permanent partners with their wives – Marilena Pollicino, Branwen Common, Val Perman and Hong Lin – despite the increasing burdens of academic life and textbook preparation. Once again, we are grateful to our wives for their help and encouragement.

It would be wrong of us not to express once again our debt to Chris Harrison (now at Cambridge University Press) for his excellence in all aspects of commissioning, editing and providing general support for the first two editions of this book. We know he remains interested in its success. Annette Abel has edited the manuscript with diligence and professionalism, correcting many of our errors and improving the transparency and readability of the text. For this we are very grateful. The staff at Pearson Higher Education, particularly Kate Brewin, Robin Lupton, Carole Drummond, Mary Lince and Gemma Papageorgiou have, as always, been helpful, enthusiastic and professional.

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YUE MA
MICHAEL COMMON
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MARCH 2011

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It is impossible to fully acknowledge our debt to the many individuals who have developed, and fostered awareness of, the discipline of natural resource and environmental economics. We hope that this debt becomes clear to the reader of this text. What will not be evident to the reader is the debt which the authors owe to those teachers who have cultivated our interests in this area of study.

Wherever the authors have drawn heavily on expositions, in written or other form, of particular individuals or organisations, care was taken to ensure that proper acknowledgement was made at the appropriate places in the text. As noted in the Preface to the Second Edition, Jack Pezzey wrote the first of the appendices to Chapter 19.

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Policy Makers, *IPCC*, Figures 10.4 and 10.29, IPCC, www.ipcc.ch; Figures 9.16 and 9.17 from Working Group III Report 'Mitigation of Climate Change', *IPCC Fourth Assessment Report: Climate Change 2007 (AR4)*, Figure 3.17, p. 199 (2007), Source: IPCC, www.ipcc.ch; Figure 9.19 from 'Working Group III, Synthesis Report for Policy Makers', *IPCC Fourth Assessment Report: Climate Change 2007 (AR4)*, Figure SPM.8 (2007), Source: IPCC, www.ipcc.ch; Figure 17.9 adapted from *The Handbook of Environmental Economics*, Blackwell (Bromley, D.W. (ed.), 1995) copyright © Blackwell, 1995. Reproduced with permission of Blackwell Publishing Ltd; and Figures 19.3, 19.4 from 'Adjusted Net Savings', The World Bank, February 2009, www.worldbank.org. Source: International Bank for Reconstruction and Development/The World Bank.

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Reproduced by permission of American Economic Association; Table 6.5 adapted from 'Costs of alternative policies for the control of nitrogen dioxide in Baltimore, Maryland', *Journal of Environmental Economics and Management*, vol. 13 (2), Tables II & III pp. 189–97 (Krupnick, A.J., 1986). Copyright © 1986, Elsevier; Table 8.2 from 'A Pilot Study on the Construction of a Scottish Environmental Input–Output System', *Report to Scottish Enterprise, Department of Economics*, University of Strathclyde, Glasgow (McNicoll, I.H. and Blackmore, D., 1993), copyright © Iain McNicoll, University of Strathclyde, Glasgow; Tables 8.3, 8.7, 8.8 adapted from 'Economic modelling and Australian carbon dioxide emissions', *Mathematics and Computers in Simulation*, vol. 33 (5–6), pp. 581–96 (Common, M.S. and Salma, U., 1992). 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Notation

List of variables

As far as possible, in using letters or symbols to denote variables or other quantities of interest, we have tried to use each character consistently to refer to one variable or quantity. This has not always been possible or desirable, however, because of the large number of variables used in the book. In the following listing, we state the meaning that is usually attached to each letter or symbol used in this way. On the few occasions where a symbol is used in two ways, this is also indicated. Where usage differs from those given below, this is made clear in the text.

A	Pollution stock (or ambient pollution level)
B	Gross benefit of an activity
C	Consumption flow *or* total cost of production of a good
D	Damage flow
E	An index of environmental pressure
e	Natural exponent
F	Reduction in pollution stock brought about by clean-up
G	Total extraction cost of a resource *or* biological growth of a resource
H	Renewable resource harvest rate
I	Investment flow
i	Market rate of interest
K	Capital stock (human-made)
L	Labour service flow
M	Emissions (pollution) flow
MP_K	Marginal product of capital
MP_L	Marginal product of labour
MP_R	Marginal product of resource
MU	Marginal utility
MU_X	Marginal utility of good X
NB	Net benefit of an activity
P	Unit price of resource (usually upper-case for gross and lower-case for net)
Q	Aggregate output flow
R	Resource extraction or use flow
r	Consumption discount rate
S	Resource stock
T	Terminal time of a planning period

t	A period or instant of time
U	Utility flow
V	Environmental clean-up expenditure
W	Social welfare flow
Z	Pollution abatement flow
δ	Social rate of return on capital
α	Pollution stock decay rate
ρ	Rate of utility time preference (utility discount rate)

The Greek characters μ, χ and ω are used for shadow prices deriving from optimisation problems.

The symbols X and Y are used in a variety of different ways in the text, depending on the context in question.

Mathematical notation

Where we are considering a function of a single variable such as

$$Y = Y(X)$$

then we write the first derivative in one of the following four equivalent ways:

$$\frac{\mathrm{d}Y}{\mathrm{d}X} = \frac{\mathrm{d}Y(X)}{\mathrm{d}X} = Y'(X) = Y_X$$

Each of these denotes the first derivative of Y with respect to X. In any particular exposition, we choose the form that seems best to convey meaning.

Where we are considering a function of several variables such as the following function of two variables:

$$Z = Z(P, Q)$$

we write first partial derivatives in one of the following equivalent ways:

$$\frac{\partial Z}{\partial P} = \frac{\partial Z(P, Q)}{\partial P} = Z_P$$

each of which is the partial derivative of Z with respect to the variable P.

We frequently use derivatives of variables with respect to time. For example, in the case of the variable S being a function of time, t, the derivative is written in one of the following equivalent forms:

$$\frac{\mathrm{d}S}{\mathrm{d}t} = \frac{\mathrm{d}S(t)}{\mathrm{d}t} = \dot{S}$$

Our most common usage is that of dot notation, as in the last term in the equalities above.

Finally, much (but not all) of the mathematical analysis in this text is set in terms of continuous time (rather than discrete time). For reasons of compactness and brevity, we chose in the first, second and third editions to avoid using the more conventional continuous-time notation $x(t)$ and to use instead the form x_t. That convention is continued here. This does, of course, run the risk of ambiguity. However, we have made every effort in the text to make explicit when discrete-time (rather than continuous-time) arguments are being used.

Introduction

Who is this book for?

This book is directed at students of economics, undertaking a specialist course in resource and/or environmental economics. Its primary use is expected to be as a principal textbook in upper-level undergraduate (final year) and taught masters-level postgraduate programmes. However, it will also serve as a main or supporting text for second-year courses (or third-year courses on four-year degree programmes) that have a substantial environmental economics component.

This fourth edition of the text is intended to be comprehensive and contemporary. It deals with all major areas of natural resource and environmental economics. The subject is presented in a way that gives a more rigorous grounding in economic analysis than is common in existing texts at this level. It has been structured to achieve a balance of theory, applications and examples, which is appropriate to a text of this level, and which will be, for most readers, their first systematic analysis of natural resource and environmental economics.

Assumptions we make about the readers of this text

We assume that the reader has a firm grasp of the economic principles covered in the first year of a typical undergraduate economics programme. In particular, it is expected that the reader has a reasonable grounding in microeconomics. However, little knowledge of macroeconomics is necessary for using this textbook. We make extensive use throughout the book of welfare economics. This is often covered in second-year micro courses, and those readers who have previously studied this will find it useful. However, the authors have written the text so that relevant welfare economics theory is developed and explained as the reader goes through the early chapters.

The authors have also assumed that the reader will have a basic knowledge of algebra. The text has been organised so that Parts I to III inclusive (Chapters 1 to 13) make use of calculus only to an elementary level. Part IV (Chapters 14 to 19) deals with the use of environmental resources over time, and so necessarily makes use of some more advanced techniques associated with dynamic optimisation. However, we have been careful to make the text generally accessible, and not to put impediments in the way of those students without substantial mathematics training. To this end, the main presentations of arguments are verbal and intuitive, using graphs as appropriate. Proofs and derivations, where these are thought necessary, are placed in appendices, available from the text's Companion Website. These can be omitted without loss of continuity, or can be revisited in a later reading.

Nevertheless, the authors believe that some mathematical techniques are sufficiently important to an economic analysis of environmental issues at this level to warrant a brief 'first-principles' exposition. We have made available, as freely downloadable documents from the Companion Website, appendices explaining the Lagrange multiplier technique of solving constrained optimisation problems, an exposition of dynamic optimisation and optimal control theory, and a brief primer on elementary matrix algebra.

Contents

As with the third edition, the new fourth edition divides the text into four parts; these parts cluster together the principal areas of interest, research and learning in natural resource and environmental economics.

Part I deals with the foundations of resource and environmental economics

The first chapter provides a background to the study of resource and environmental economics by putting the field in its context in the history of economics, and by briefly outlining the fundamental characteristics of an economics approach to environmental analysis. The text then, in Chapter 2, considers the origins of the sustainability problem by discussing economy–environment interdependence, introducing some principles from environmental science, and by investigating the drivers of environmental impact. Sustainable development is intrinsically related to the quality of human existence, and we review here some of the salient features on the current state of human development. Chapter 3 examines the ethical underpinnings of resource and environmental economics. Part I finishes, in Chapter 4, with a comprehensive review of the theory of static welfare economics, and provides the fundamental economic tools that will be used throughout the rest of the book.

Part II covers what is usually thought to be 'environmental economics'

A principal focus of the six chapters in Part II is the analysis of pollution. We deal here with pollution targets, in Chapter 5, and with methods of attaining pollution targets (that is, instruments), in Chapter 6. We are careful to pay proper attention to the limits of economic analysis in these areas. Pollution policy is beset by problems of limited information and uncertainty, and Chapter 7 is entirely devoted to this matter. Many environmental problems spill over national boundaries, and can only be successfully dealt with by means of international cooperation. Again, we regard this topic of sufficient importance to warrant a chapter, 9, devoted to it. A central feature of this chapter is our use of game theory as the principal tool by which we study the extent and evolution of international cooperation on environmental problems. The spatial incidence of, and perhaps also the aggregate amount of, environmental pollution is affected by the processes associated with international trade; trade and the environment is covered in Chapter 10. Finally, the authors stress the limits of partial equilibrium analysis, and correspondingly the advantages of using system-wide economic analysis. In Chapter 8 we take the reader through the two principal tools of economy-wide economy–environment modelling, input–output analysis and computable general equilibrium modelling. The ways in which general equilibrium – as opposed to partial equilibrium – modelling can enhance our understanding of resource and environmental issues and provide a rich basis for policy analysis are demonstrated here.

Part III is concerned with the principles and practice of project appraisal

Many practitioners will find that their work involves making recommendations about the desirability of particular projects. Cost–benefit analysis is the central tool developed by economists to support this activity. We provide, in Chapter 11, a careful summary of that technique, paying close attention to its theoretical foundations in intertemporal welfare economics. Our exposition also addresses the limits – in principle and in practice – of cost–benefit analysis, and outlines some other approaches to project appraisal, including multi-criteria analysis. A distinguishing characteristic of the economic approach to project appraisal is its insistence on the evaluation of environmental impacts on a basis that allows comparability with the other costs and benefits of the project. In Chapter 12 we examine the economic theory and practice of valuing environmental (and other non-marketed) services, giving examples of the application of each of the more commonly used methods. Inevitably, decisions are made within a setting of risk and uncertainty, and in which actions will often entail irreversible consequences. Chapter 13 examines how these considerations might shape the ways in which projects should be appraised.

Part IV covers what is commonly known as resource economics

The basic economic approach to natural resource exploitation is set out in Chapter 14. In Chapter 15 we focus on non-renewable resources, while Chapter 17 is about the economics of renewable resource harvesting and management, focusing especially on ocean fisheries. Forest resources have some special characteristics, and are the subject of Chapter 18. Chapter 16 revisits the analysis of pollution problems, but this time focusing on stock pollutants, where the analytical methods used to study resources are applicable. In this chapter pollution generation is linked to the extraction and use of natural resources, as is necessary in order to develop a sound understanding of many environmental problems, in particular that of the enhanced greenhouse effect. Finally, Chapter 19 returns to the question of sustainability in the context of a discussion of the theory and practice of environmental accounting.

Perspectives

All books look at their subject matter through one or more 'lenses' and this one is no exception.

- It adopts an *economics* perspective, while nevertheless recognising the limits of a purely economic analysis and the contributions played by other disciplines.
- It is an *environmental* economics (as opposed to an *ecological* economics) text, although the reader will discover something here of what an ecological economics perspective entails.
- The authors have oriented the text around the organising principles of *efficiency* and *sustainability*.
- Many textbook expositions fail to distinguish properly between the notions of efficiency and optimality; it is important to use these related, but nevertheless separate, ideas properly.
- Although the partitioning of the text might be taken to imply a separation of resource economics from environmental economics, our treatment of topics has made every effort to avoid this.
- Some topics and ideas appear at several points in the book, and so are examined from different perspectives and in various contexts. Examples include the Hartwick rule and the safe minimum standard principle.
- Substantial attention is given to the consequences of limited information (or uncertainty) for policy making.

The textbook as a learning resource

The authors are aware that students need a variety of resources for effective learning. Following developments in the previous edition, we have tried to move this fourth edition of the text closer to providing a full set of such resources. This has been done mainly through the development of an accompanying website. Having such a website allows the authors to update content, examples, web link addresses, and so on in a way that is not possible with a printed textbook itself.

The content of that site is described at length in the section on *Additional Resources*. At this point it is sufficient just to note that these consist (principally but not exclusively) of

- appendices to the chapters in the textbook itself. All but five of the chapters in this book have associated with them one or more appendices, amounting to 33 appendices in all. Each of these is a substantial document, often giving a more formal or mathematical treatment of a technique or theme covered in the chapters of the text.
- an extensive set of web links, carefully structured to facilitate further reading and research;
- specimen answers for the Discussion Questions and Problems that appear at the end of the chapters in the book;
- many additional online Word documents, examining at greater length some topics that had relatively brief coverage in the main text (such as biodiversity, agriculture, traffic);
- a large number of Excel files that use simulation techniques to explore environmental issues, problems, or policies. These can be used by the reader to enhance understanding through exploring a topic further; and teachers may work them up into problems that give powerful insight.
- Maple files that one of the authors uses extensively for teaching and research purposes.

Many of our readers will have access to this software package through a university or college with which they are associated. But even where a reader does not have access to Maple software, readable (rich text format) versions of the Maple files will be available, and the structure and techniques used to carry out simulations or to explore environmental issues, problems, or policies will be evident from them. With a little ingenuity, a reader without Maple access should be able to reproduce these exercises using software that can substitute for Maple (such as Mathematica, or one of the freely distributed, public-domain packages such as Maxima).

Other pedagogical features

We have gone to some trouble to use, as far as is possible, consistent notation throughout the book. A list of the main symbols used and their usual meanings is given on pages xx–xxii. However, given the range of material covered it has not been possible to maintain a full one-to-one correspondence between symbols and referents throughout the book. Some symbols do have different meanings in different places. Wherever there is the possibility of confusion we have made explicit what the symbols used mean at that point in the text.

Secondly, each chapter begins with learning objectives and concludes with a chapter summary. While these are relatively modest in extent, we hope the reader will nevertheless find them useful. Finally, each chapter also contains a guide to further reading. Several of these are very extensive. Combined with the website-based links and bibliographies, the reader will find many pointers on where to go next.

Course designs

The authors do, of course, hope that this text will be used for a full course of study involving the material in all 19 chapters. However, we are aware that this would be time-consuming and may not fit with all institutional structures. We therefore offer the following three suggestions as to how the text might be used for shorter courses. Suggestions A and B avoid the chapters where dynamic optimisation techniques need to be used, but still include material on sustainability and the principles and application of cost–benefit analysis. In all cases, courses could be further shortened for students with a strong economics background by treating some parts, at least, of Chapters 4 and 11 as revision material. We do not recommend that this material be completely dropped for any course. Obviously, other permutations are also possible.

A: An environmental economics course

Part I Foundations

Chapter 1 An introduction to natural resource and environmental economics
Chapter 2 The origins of the sustainability problem
Chapter 3 Ethics, economics and the environment
Chapter 4 Welfare economics and the environment

Part II Environmental Pollution

Chapter 5 Pollution control: targets
Chapter 6 Pollution control: instruments
Chapter 9 International environmental problems
Chapter 10 Trade and the environment

Part III Project Appraisal

Chapter 11 Cost–benefit analysis
Chapter 12 Valuing the environment

B: An environmental policy course

Part I Foundations

Chapter 2 The origins of the sustainability problem
Chapter 3 Ethics, economics and the environment
Chapter 4 Welfare economics and the environment

Part II Environmental Pollution

Chapter 5 Pollution control: targets
Chapter 6 Pollution control: instruments
Chapter 7 Pollution policy with imperfect information
Chapter 8 Economy-wide modelling
Chapter 9 International environmental problems
Chapter 10 Trade and the environment

Part III Project Appraisal

Chapter 11 Cost–benefit analysis
Chapter 12 Valuing the environment
Chapter 13 Irreversibility, risk and uncertainty

C: A resource economics and policy course

Part I Foundations

Chapter 2 The origins of the sustainability problem
Chapter 4 Welfare economics and the environment
Chapter 8 Economy-wide modelling

Part III Project Appraisal

Chapter 11 Cost–benefit analysis
Chapter 12 Valuing the environment
Chapter 13 Irreversibility, risk and uncertainty

Part IV Natural Resource Exploitation

Chapter 14 The efficient and optimal use of natural resources
Chapter 15 The theory of optimal resource extraction: non-renewable resources
Chapter 16 Stock pollution problems
Chapter 17 Renewable resources
Chapter 18 Forest resources
Chapter 19 Accounting for the environment

Additional resources

On the back cover of this textbook, you will find the URL (website address) of a site that is available to accompany the text. For convenience, we reproduce the web address again here; it is www.pearsoned.co.uk/perman.

Clicking on this hyperlink will take you to a page on the Pearson website that provides information about Supporting Resources for this textbook. The contents of that page are listed on page i, more fully on pages xxvi–xxvii, and at the foot of this page.

Readers are also encouraged to visit the author's own textbook website at http://personal.strath.ac.uk/r.perman/menu.htm. This accompanying author's website will undergo a process of evolution throughout the life of the textbook. Periodically, the content of the web pages will be reviewed and updated. As errata become known to us, the relevant author's web pages will be periodically updated.

The authors welcome suggestions for further items to include on these web pages. If you would like to make any such suggestion, or if you have a particular 'ready-made' item that you feel would be a useful addition, please e-mail Roger Perman at r.perman@strath.ac.uk. The authors will consider these suggestions carefully and, wherever possible and desirable, incorporate them (with proper attribution) in these web pages.

Supporting resources

Visit **www.pearsoned.co.uk/perman** to find valuable online resources

Companion Website for students

- Additional materials to enhance your knowledge
- Excel files that use simulations techniques to explore environmental issues, problems and policies
- Maple examples and spreadsheet exercises to practise and test your understanding
- Appendices

For instructors

- Complete, downloadable Instructor's Manual
- Answers to questions in the text
- PowerPoint slides that can be downloaded and used for presentations

For more information please contact your local Pearson Education sales representative or visit **www.pearsoned.co.uk/perman**

We now give a little more information about some of the textbook's web-based supporting resources.

Appendices

These are available on the accompanying website but are not in the textbook itself.

- 3.1 The Lagrange multiplier method of solving constrained optimisation problems
- 3.2 Social welfare maximisation
- 4.1 Conditions for efficiency and optimality
- 4.2 Market outcomes
- 4.3 Market failure
- 5.1 Matrix algebra
- 5.2 Spatially differentiated stock pollution: a numerical example
- 6.1 The least-cost theorem and pollution control instruments
- 8.1 A general framework for environmental input–output analysis
- 8.2 The algebra of the two-sector CGE model
- 9.1 Some algebra of international treaties
- 11.1 Conditions for intertemporal efficiency and optimality
- 11.2 Markets and intertemporal allocation
- 12.1 Demand dependency
- 12.2 Weak complementarity with observable Marshallian demand curves
- 13.1 Irreversibility and development: future known
- 13.2 Irreversibility, development and risk
- 14.1 The optimal control problem and its solution using the maximum principle
- 14.2 The optimal solution to the simple exhaustible resource depletion problem
- 14.3 Optimal and efficient extraction or harvesting of a renewable or non-renewable resource in the presence of resource extraction costs
- 15.1 Solution of the multi-period resource depletion model
- 15.2 The monopolist's profit-maximising extraction programme
- 15.3 A worked numerical example
- 17.1 The discrete-time analogue of the continuous time fishery models examined in Chapter 17
- 17.2 Derivation of steady-state equilibrium for an open access fishery and for a private property fishery
- 17.3 The dynamics of renewable resource harvesting
- 18.1 Mathematical derivations
- 18.2 The length of a forest rotation in the infinite rotation model: some comparative statics
- 19.1 National income, the return on wealth, Hartwick's rule and sustainable income
- 19.2 Adjusting national income measurement to account for the environment
- 19.3 Theory for an imperfect economy
- 19.4 The UNSTAT proposals
- 19.5 El Serafy's method for the estimation of the depreciation of natural capital

This set of resources contains downloadable (pdf and Word) versions of the full set of Appendices for the various chapters of the textbook itself. Given that depth of understanding is intrinsically desirable, we would urge all readers of this book to read and study these appendices. Nevertheless, all of the appendices are technical elaborations on matters covered more intuitively in the chapters of the textbook, and the reader should find that his or her ability to understand the content of the text is not dependent on being able to fully grasp appendix material.

Additional Materials

This is intended by the authors to be a very significant resource available to readers. It consists of a set of documents that delve more deeply into some aspects of material covered (perhaps only briefly) in the book itself, or which provide more information on the policy or institutional facets of issues in natural resource and environmental economics, or which allow the reader (or an instructor) to undertake model simulations or carry our comparative static or comparative dynamic analysis. These additional materials are intended to be entirely optional and genuinely additional. It is not necessary for the reader to read, study, or work through any of them. It is not required that you use any of these materials to follow any of the arguments and/or examples used in the text. The textbook has been written in such a way that it stands alone, and does not intrinsically depend on these additional materials. (Where we felt something was necessary, it was included in the main text.)

However, the fact that we have included these materials does imply that the authors think you may find some of them useful. Some materials are designed to broaden knowledge (by giving, in Word files, additional commentary on related matters). Others are aimed at deepening understanding by using standard software packages (such as Excel) to show how numerical examples used in the text were obtained, and to allow the reader to experiment a little, perhaps by changing parameter values from those used in the text and observing what happens. Occasionally we use the symbolic mathematical package Maple for some of the items in *Additional Materials*. Many readers will be unfamiliar with this package, and you should not worry if they are not, therefore, useful to you. But please note that Maple is increasingly being used in higher education, is not difficult to learn, and can be a very powerful tool to have at your disposal. You may wish to follow some of our suggestions about learning how to use this package.

Finally, we also anticipate that some lecturers and instructors will wish to adapt some of these materials for class use (much as many of the files you will find here benefit from other writers' earlier work). The authors believe that much useful learning can take place if instructors adapt some of the spreadsheet exercises as exploratory problems and set them as individual or group tasks for their students.

Accessing the Additional Materials

Most of the chapters in the textbook refer to one or more files that are called *Additional Materials*, plus a specific file name.

Some of the *Additional Materials* are available from the Pearson website; others are available from the author's own web pages.

Answers to Questions in the Text

All chapters in this textbook (except the first) contain a small number of Discussion Questions and Problems. Answers to most of these are available. Those answers are collated chapter by chapter, and can be accessed through the main table that you will find on this web page.

Environmental Economics Links

As we remarked in the third edition, a huge volume of information of interest to the environmental economist can now be found on the Internet. This can be read online, printed for future reference or saved to disk. It is hardly a novel idea to compile a set of 'Useful links' and to place this on one's own website. We have also done that. You will find these web links on the author's own website at http://personal.strath.ac.uk/r.perman/menu.htm.

However, we have reasons for believing you may find this one more useful than most. The main reason for this belief lies in its structure. Actually, these links are structured in two different ways:

- by chapter topic;
- by the provider type.

For example, suppose that you have just read Chapter 17 (on renewable resources) and wish to be pointed to a set of web links that are particularly useful in relation to the content of that chapter. Then go to the chapter-by-chapter menu option, select 17 from the table, and the links will be provided. We do not claim that our classification is always uncontentious; but the authors have tried to be helpful. Some of the web links contain brief annotated commentary that may help you select more efficiently.

The 'By organisation link' structure is more conventional but still very useful, given that so much of value comes from a relatively small set of organisations. You will find that we have further sub-classified this set in various ways to help your searching. It will be too cumbersome to explain the classification structure here. It will be much simpler for you to follow the appropriate link from the Main Menu and view it directly. You will no doubt know already many incredibly good Internet sites maintained by organisations with an interest in the environment (such as those of the US EPA, various United Nations bodies, and many environmental ministries). You may be less aware of the existence of a large number of excellent university or research group sites, or those of various individuals and non-governmental organisations (NGOs).

Note also that the main menu has one specific item labelled as 'A variety of Bibliographies'. Listed here are not only links to some excellent printed book and/or article bibliographies but also links to a small number of other exceptionally good website compilations. You do not have to rely only on us, therefore!

We are always looking for new suggestions for links to be included in our lists. Please e-mail suggestions to Roger Perman (address given earlier).

Site availability

The URL for the accompanying companion website maintained by Pearson is www.pearsoned.co.uk/perman. However, some of the materials that are associated with this textbook are updated on a regular basis and are only to be found on Roger Perman's personal website for this textbook, on a server located at the University of Strathclyde. This can be accessed via the URL http://personal.strath.ac.uk/r.perman/menu.htm. In common with web addresses at many university servers, this address may change at some time in the future. In the event of such a change, a link to the revised address will be given on the accompanying website.

PART I Foundations

CHAPTER 1 An introduction to natural resource and environmental economics

Contemplation of the world's disappearing supplies of minerals, forests, and other exhaustible assets has led to demands for regulation of their exploitation. The feeling that these products are now too cheap for the good of future generations, that they are being selfishly exploited at too rapid a rate, and that in consequence of their excessive cheapness they are being produced and consumed wastefully has given rise to the conservation movement.

Hotelling (1931)

Learning objectives

In this chapter you will

- be introduced to the concepts of efficiency, optimality and sustainability
- learn about the history of natural resource and environmental economics
- have the main issues of modern resource and environmental economics identified
- see an overview and outline of the structure of this text

Introduction

The three themes that run through this book are efficiency, optimality and sustainability. In this chapter we briefly explain these themes, and then look at the emergence of the field of study which is the economic analysis of natural resources and the environment. We then identify some of the key features of that field of study, and indicate where, later in the book, the matters raised here are discussed more fully.

1.1 Three themes

The concepts of efficiency and optimality are used in specific ways in economic analysis. We will be discussing this at some length in Chapter 4. However, a brief intuitive account here will be useful. One way of thinking about efficiency is in terms of missed opportunities. If resource use is wasteful in some way then opportunities are being squandered; eliminating that waste (or inefficiency) can bring net benefits to some group of people. An example is energy inefficiency. It is often argued that much energy is produced or used inefficiently, and that if different techniques were employed significant resource savings could be gained with no loss in terms of final output.

This kind of argument usually refers to some kind of technical or physical inefficiency. Economists usually assume away this kind of inefficiency, and focus on allocative inefficiencies. Even where resources are used in technically efficient ways, net benefits are sometimes squandered. For example, suppose that electricity can be, in technically efficient ways, generated by the burning of either some heavily polluting fossil fuel, such as coal, or a less polluting alternative fossil fuel, such as gas. Because of a lower price for the former fuel, it is chosen by profit-maximising electricity producers. However, the pollution results in damages which necessitate expenditure on health care and clean-up operations. These expenditures, not borne by the electricity supplier, may exceed the cost saving that electricity producers obtain from using coal.

If this happens there is an inefficiency that results from resource allocation choices even where there are no technical inefficiencies. Society as a whole would obtain positive net benefits if the less polluting alternative were used. We show throughout the book that such allocative inefficiencies will be pervasive in the use of natural and environmental resources in pure market economies. A substantial part of environmental economics is concerned with how economies might avoid inefficiencies in the allocation and use of natural and environmental resources.

The second concept – optimality – is related to efficiency, but is distinct from it. To understand the idea of optimality we need to have in mind:

1. a group of people taken to be the relevant 'society';
2. some overall objective that this society has, and in terms of which we can measure the extent to which some resource-use decision is desirable from that society's point of view.

Then a resource-use choice is socially optimal if it maximises that objective given any relevant constraints that may be operating.

As we shall see (particularly in Chapter 4), the reason efficiency and optimality are related is that it turns out to be the case that a resource allocation cannot be optimal unless it is efficient. That is, efficiency is a necessary condition for optimality. This should be intuitively obvious: if society squanders opportunities, then it cannot be maximising its objective (whatever that might be). However, efficiency is not a sufficient condition for optimality; in other words, even if a resource allocation is efficient, it may not be socially optimal. This arises because there will almost always be a multiplicity of different efficient resource allocations, but only one of those will be 'best' from a social point of view. Not surprisingly, the idea of optimality also plays a role in economic analysis.

The third theme is sustainability. For the moment we can say that sustainability involves taking care of posterity. Why this is something that we need to consider in the context of resource and environmental economics is something that we will discuss in the next chapter. Exactly what 'taking care of posterity' might mean is discussed in Chapter 3. On first thinking about this, you might suspect that, given optimality, a concept such as sustainability is redundant. If an allocation of resources is socially optimal, then surely it must also be sustainable? If sustainability matters, then presumably it would enter into the list of society's objectives and would get taken care of in achieving optimality. Things are not quite so straightforward. The pursuit of optimality as usually considered in economics will not necessarily take adequate care of posterity. If taking care of posterity is seen as a moral obligation, then the pursuit of optimality as economists usually specify it will need to be constrained by a sustainability requirement.

1.2 The emergence of resource and environmental economics

We now briefly examine the development of resource and environmental economics from the time of the industrial revolution in Europe.

1.2.1 Classical economics: the contributions of Smith, Malthus, Ricardo and Mill to the development of natural resource economics

While the emergence of natural resource and environmental economics as a distinct sub-discipline has been a relatively recent event, concern with the substance of natural resource and environmental issues has much earlier antecedents. It is evident, for example, in the writings of the classical economists, for whom it was a major concern. The label 'classical' identifies a number of economists writing in the eighteenth and nineteenth centuries, a period during which the industrial revolution was taking place (at least in much of Europe and North America) and agricultural productivity was growing rapidly. A recurring theme of political–economic debate concerned the appropriate institutional arrangements for the development of trade and growth.

These issues are central to the work of Adam Smith (1723–1790). Smith was the first writer to systematise the argument for the importance of markets in allocating resources, although his emphasis was placed on what we would now call the dynamic effects of markets. His major work, *An Inquiry into the Nature and Causes of the Wealth of Nations* (1776), contains the famous statement of the role of the 'invisible hand':

> But it is only for the sake of profit that any man employs a capital in the support of industry; and he will always, therefore, endeavour to employ it in the support of that industry of which the produce is likely to be of the greatest value, or to exchange for the greatest quantity, either of money or of other goods.
>
> As every individual, therefore, endeavours as much as he can both to employ his capital in the support of domestic industry, and so to direct that industry that its produce may be of the greatest value; every individual necessarily labours to render the annual revenue of the society as great as he can. He generally, indeed, neither intends to promote the public interest, nor knows how much he is promoting it . . . he is, in this as in many other cases, led by an invisible hand to promote an end which was no part of his intention . . .
>
> . . . By pursuing his own interest he frequently promotes that of society more effectively than when he really intends to promote it.
>
> *Smith ([1776] 1961), Book IV, Ch. 2, p. 477*

This belief in the efficacy of the market mechanism is a fundamental organising principle of the policy prescriptions of modern economics, including resource and environmental economics, as will be seen in our account of it in the rest of the book.

A central interest of the classical economists was the question of what determined standards of living and economic growth. Natural resources were seen as important determinants of national wealth and its growth. Land (sometimes used to refer to natural resources in general) was viewed as limited in its availability. When to this were added the assumptions that land was a necessary input to production and that it exhibited diminishing returns, the early classical economists came to the conclusion that economic progress would be a transient feature of history. They saw the inevitability of an eventual stationary state, in which the prospects for the living standard of the majority of people were bleak.

This thesis is most strongly associated with Thomas Malthus (1766–1834), who argued it most forcefully in his *Essay on the Principle of Population* (1798), giving rise to the practice of describing those who now question the feasibility of continuing long-run economic growth as 'neo-Malthusian'. For Malthus, a fixed land quantity, an assumed tendency for continual positive population growth, and diminishing returns in agriculture implied a tendency for output per capita to fall over time. There was, according to Malthus, a long-run tendency for the living standards of the mass of people to be driven down to a subsistence level. At the subsistence wage level, living standards would be such that the population could just reproduce itself, and the economy would attain a steady state with a constant population size and constant, subsistence-level, living standards.

This notion of a steady state was formalised and extended by David Ricardo (1772–1823), particularly in his *Principles of Political Economy and Taxation* (1817). Malthus's assumption of a fixed stock of land was replaced by a conception in which land was available in parcels of varying quality. Agricultural output could be expanded by increasing the intensive margin (exploiting a given parcel of land more intensively) or by increasing the extensive margin (bringing previously uncultivated land into productive use). However, in either case, returns to the land input were taken to be diminishing. Economic development then proceeds in such a way that the 'economic surplus' is appropriated increasingly in the form of rent, the return to land, and development again converges toward a Malthusian stationary state.

In the writings of John Stuart Mill (1806–1873) (see in particular Mill (1857)) one finds a full statement of classical economics at its culmination. Mill's work utilises the idea of diminishing returns, but recognises the countervailing influence of the growth of knowledge and technical progress in agriculture and in production more generally. Writing in Britain when output per person was apparently rising, not falling, he placed less emphasis on diminishing returns, reflecting the relaxation of the constraints of the extensive margin as colonial exploitation opened up new tranches of land, as fossil fuels were increasingly exploited, and as innovation rapidly increased agricultural productivity. The concept of a stationary state was not abandoned, but it was thought to be one in which a relatively high level of material prosperity would be attained.

Foreshadowing later developments in environmental economics, and the thinking of conservationists, Mill adopted a broader view of the roles played by natural resources than his predecessors. In addition to agricultural and extractive uses of land, Mill saw it as a source of amenity values (such as the intrinsic beauty of countryside) that would become of increasing relative importance as material conditions improved. We discuss a modern version of this idea in Chapter 11.

Mill's views are clearly revealed in the following extract from his major work.

> Those who do not accept the present very early stage of human improvement as its ultimate type may be excused for being comparatively indifferent to the kind of economic progress which excites the congratulations of ordinary politicians: the mere increase of production . . . It is only in the backward countries of the world that increased production is still an important object; in those most advanced, what is needed is a better distribution . . . There is room in the world, no doubt, and even in old countries, for a great increase in population, supposing the arts of life to go on improving, and capital to increase. But even if innocuous, I confess I see very little reason for desiring it. The density of population necessary to enable mankind to obtain, in the greatest degree, all of the advantages both of cooperation and of social intercourse, has, in all the most populous countries, been attained. A population may be too crowded, though all be amply supplied with food and raiment. It is not good for man to be kept perforce at all times in the presence of his species . . . Nor is there much satisfaction in contemplating the world with nothing left to the spontaneous activity of nature: with every rood of land brought into cultivation, which is capable of growing food for human beings; every flowery waste or natural pasture ploughed up, all quadrupeds or birds which are not domesticated for man's use exterminated as his rivals for food, every hedgerow or superfluous tree rooted out, and scarcely a place left where a wild shrub or flower could grow without being eradicated as a weed in the name of improved agriculture. If the earth must lose that great portion of its pleasantness which it owes to things that the unlimited increase of wealth and population would extirpate from it, for the mere purpose of enabling it to support a larger, but not a happier or better population, I sincerely hope, for the sake of posterity, that they will be content to be stationary long before necessity compels them to it.
>
> *Mill (1857), Book IV*

It is worth noting explicitly that at the time that Mill wrote this the global population was less than one quarter of what it is now, and that average per capita income, as gross domestic product (GDP), in the then rich parts of the world was of the order of 10% of what it is now.[1] We briefly review some of the recent research on the determinants of self-assessed individual human happiness in Chapter 3.

1.2.2 Neoclassical economics: marginal theory and value

A series of major works published in the 1870s began the replacement of classical economics by what subsequently became known as 'neoclassical economics'. One outcome of this was a change in the manner in which value was explained. Classical economics saw value as arising from the labour power embodied (directly and indirectly) in output, a view which found its fullest embodiment in the work of Karl Marx. Neoclassical economists explained value as being determined in exchange, so reflecting preferences and costs of production. The concepts of price and value ceased to be distinct. Moreover, previous notions of absolute scarcity and value were replaced by a concept of relative scarcity, with relative values (prices) determined by the forces of supply and demand. This change in emphasis paved the way for the development of welfare economics, to be discussed shortly.

At the methodological level, the technique of marginal analysis was adopted, allowing earlier notions of diminishing returns to be given a formal basis in terms of diminishing marginal productivity in the context of an explicit production function. Jevons (1835–1882) and Menger (1840–1921) formalised the theory of consumer preferences in terms of utility and demand theory. The evolution of

[1] These statements are based on estimates in Table 2.1 in Maddison (2007). It gives world population as 1272 million in 1870 and 6279 million in 2003, and per capita GDP for Western Europe as 1960 1990$s in 1870 and 19 912 1990$s in 2003.

neoclassical economic analysis led to an emphasis on the structure of economic activity, and its allocative efficiency, rather than on the aggregate level of economic activity. Concern with the prospects for continuing economic growth receded, perhaps reflecting the apparent inevitability of growth in Western Europe at this time. Leon Walras (1834–1910) developed neoclassical General Equilibrium Theory, and in so doing provided a rigorous foundation for the concepts of efficiency and optimality that we employ extensively in this text. Alfred Marshall (1842–1924) (see *Principles of Economics*, 1890) was responsible for elaboration of the partial equilibrium supply-and-demand-based analysis of price determination so familiar to students of modern microeconomics. A substantial part of modern environmental economics continues to use these techniques as tools of exposition, as do we at many points throughout the book.

We remarked earlier that concern with the level (and the growth) of economic activity had been largely ignored in the period during which neoclassical economics was being developed. Economic depression in the industrialised economies in the inter-war years provided the backcloth against which John Maynard Keynes (1883–1946) developed his theory of income and output determination. The Keynesian agenda switched attention to aggregate supply and demand, and the reasons why market economies may fail to achieve aggregate levels of activity that involve the use of all of the available inputs to production. Keynes was concerned to explain, and provide remedies for, the problem of persistent high levels of unemployment, or recession.

This direction of development in mainstream economics had little direct impact on the emergence of resource and environmental economics. However, Keynesian 'macroeconomics', as opposed to the microeconomics of neoclassical economics, was of indirect importance in stimulating a resurgence of interest in growth theory in the middle of the twentieth century, and the development of a neoclassical theory of economic growth. What is noticeable in early neoclassical growth models is the absence of land, or any natural resources, from the production function used in such models. Classical limits-to-growth arguments, based on a fixed land input, did not have any place in early neoclassical growth modelling.

The introduction of natural resources into neoclassical models of economic growth occurred in the 1970s, when some neoclassical economists first systematically investigated the efficient and optimal depletion of resources. This body of work, and the developments that have followed from it, is natural resource economics. The models of efficient and optimal exploitation of natural resources that we present and discuss in Chapters 14, 15, 17 and 18 are based on the writings of those authors. We will also have call to look at such models in Chapter 19, where we discuss the theory of accounting for the environment as it relates to the question of sustainability.

1.2.3 Welfare economics

The final development in mainstream economic theory that needs to be briefly addressed here is the development of a rigorous theory of welfare economics. Welfare economics, as you will see in Chapter 4, attempts to provide a framework in which normative judgements can be made about alternative configurations of economic activity. In particular, it attempts to identify circumstances under which it can be claimed that one allocation of resources is better (in some sense) than another.

Not surprisingly, it turns out to be the case that such rankings are only possible if one is prepared to accept some ethical criterion. The most commonly used ethical criterion adopted by classical and neoclassical economists derives from the utilitarian moral philosophy, developed by David Hume, Jeremy Bentham and John Stuart Mill. We explore this ethical structure in Chapter 3. Suffice to say now that utilitarianism has social welfare consisting of some weighted average of the total utility levels enjoyed by all individuals in the society.

Economists have attempted to find a method of ranking different states of the world which does not require the use of a social welfare function, and makes little use of ethical principles, but is nevertheless useful in making prescriptions about resource allocation. The notion of economic efficiency, also known as allocative efficiency or Pareto optimality (because it was developed by Vilfredo Pareto (1897)), is what they have come up with. These ideas are examined at length in Chapter 4. It can be shown that, given certain rather stringent conditions, an economy organised as a competitive market economy will attain a state of economic efficiency. This is the modern, and rigorous, version of Adam Smith's story about the benign influence of the invisible hand.

Where the conditions do not hold, markets do not attain efficiency in allocation, and a state of 'market failure' is said to exist. One manifestation of market failure is the phenomenon of 'externalities'. These are situations where, because of the structure of property rights, relationships between economic agents are not all mediated through markets. Market failure and the means for its correction will be discussed in Chapter 4.

The problem of pollution is a major concern of environmental economics. It first attracted the attention of economists as a particular example of the general class of externalities. Important early work in the analysis of externalities and market failure is to be found in Marshall (1890). The first systematic analysis of pollution as an externality is to be found in Pigou (1920). However, environmental economics did not really 'take off' until the 1970s. The modern economic treatment of problems of environmental pollution is covered in Chapters 5, 6 and 7, and in Chapter 16.

Environmental economics is also concerned with the natural environment as a source of recreational and amenity services, which is what Mill was drawing attention to in the quotation above. This role for the environment can be analysed using concepts and methods similar to those used in looking at pollution problems. This branch of modern environmental economics is covered in Chapters 11, 12 and 13. Like pollution economics, it makes extensive use of the technique of cost–benefit analysis, which emerged in the 1950s and 1960s as a practical vehicle for applied welfare economics and policy advice. The basic structure and methodology of cost–benefit analysis is dealt with in Chapter 11, building on the discussion of market failure and public policy in Chapter 4.

The modern sub-disciplines of natural resource economics and environmental economics have largely distinct roots in the core of modern mainstream economics. The former emerged mainly out of neoclassical growth economics, the latter out of welfare economics and the study of market failure. Both can be said to date effectively from the early 1970s, though of course earlier contributions can be identified.

1.2.4 Ecological economics

Ecological economics is a relatively new, interdisciplinary, field. In the 1980s a number of economists and natural scientists came to the conclusion that if progress was to be made in understanding and addressing environmental problems it was necessary to study them in an interdisciplinary way. The International Society for Ecological Economics was set up in 1989. The precise choice of name for this society may have been influenced by the fact that a majority of the natural scientists involved were ecologists, but more important was the fact that economics and ecology were seen as the two disciplines most directly concerned with what was seen as the central problem – sustainability.

Ecology is the study of the distribution and abundance of animals and plants. A central focus is an ecosystem, which is an interacting set of plant and animal populations and their abiotic, non-living, environment. The Greek word 'oikos' is the common root for the 'eco' in both economics and ecology. Oikos means 'household', and it could be said that ecology is the study of nature's housekeeping, while economics is the study of human housekeeping. Ecological economics could then be said to be the study of how these two sets of housekeeping are related to one another. Earlier in this chapter we said that sustainability involves taking care of posterity. Most of those who would wish to be known as ecological economists are concerned that the scale of human housekeeping is now such that it threatens the viability of nature's housekeeping in ways which will adversely affect future generations of humans.

The distinguishing characteristic of ecological economics is that it takes as its starting point and its central organising principle the fact that the economic system is part of the larger system that is planet earth. It starts from the recognition that the economic and environmental systems are interdependent, and studies the joint economy–environment system in the light of principles from the natural sciences, particularly thermodynamics and ecology. We shall briefly discuss these matters in the next chapter, which has the title 'The origins of the sustainability problem', as it is the interdependence of economic and natural systems that gives rise to the sustainability problem.

Kenneth Boulding is widely regarded as one of the 'founding fathers' of ecological economics. Box 1.1 summarises a paper that he wrote in 1966 which uses vivid metaphors to indicate the change in ways of thinking that he saw as necessary, given the laws of nature and their implications for economic activity. As we have seen, the dependence of economic activity on its material base – the natural environment – was a central concern of classical economics, but not of neoclassical economics. Boulding was one of a few scholars, including some economists, who continued, during the ascendancy of neoclassical economics, to insist on the central importance of studying economics in a way which takes on board what is known about the laws of nature as they affect the material basis for economic activity. As is made clear in Box 1.1, Boulding did not, and ecological economics does not, take the view that everything that resource and environmental economics has to say, for example, about using price incentives to deal with environmental problems is wrong. Rather, the point is that what it has to say needs to be put in the proper context, one where the economic system is seen as a subsystem of a larger system.

Box 1.1 Economics of 'Spaceship Earth'

In a classic paper written in 1966, 'The economics of the coming Spaceship Earth', Kenneth Boulding discusses a change in orientation that is required if mankind is to achieve a perpetually sustainable economy. He begins by describing the prevailing image which man has of himself and his environment. The 'cowboy economy' describes a state of affairs in which the typical perception of the natural environment is that of a virtually limitless plain, on which a frontier exists that can be pushed back indefinitely. This economy is an open system, involved in interchanges with the world outside. It can draw upon inputs from the outside environment, and send outputs (in the form of waste residuals and so on) to the outside. In the cowboy economy perception, no limits exist on the capacity of the outside to supply or receive energy and material flows.

Boulding points out that, in such an economy, the measures of economic success are defined in terms of flows of materials being processed or transformed. Roughly speaking, income measures such as GDP or GNP reflect the magnitudes of these flows – the cowboy perception regards it as desirable that these flows should be as large as possible.

However, Boulding argues, this economy is built around a flawed understanding of what is physically possible in the long run. A change in our perception is therefore required to one in which the earth is recognised as being a closed system or, more precisely, a system closed in all but one respect – energy inputs are received from the outside (such as solar energy flows) and energy can be lost to the outside (through radiative flows, for example). In material terms, though, planet earth is a closed system: matter cannot be created or destroyed, and the residuals from extraction, production and consumption activities will always remain with us, in one form or another.

Boulding refers to this revised perception as that of the 'spaceman economy'. Here, the earth is viewed as a single spaceship, without unlimited reserves of anything. Beyond the frontier of the spaceship itself, there exist no reserves from which the spaceship's inhabitants can draw resources nor sinks into which they can dispose of unwanted residuals. On the contrary, the spaceship is a closed material system, and energy inputs from the outside are limited to those perpetual but limited flows that can be harnessed from the outside, such as solar radiation.

Within this spaceship, if mankind is to survive indefinitely, man must find his place in a perpetually reproduced ecological cycle. Materials usage is limited to that which can be recycled in each time period; that, in turn, is limited by the quantity of solar and other external energy flows received by the spaceship.

Box 1.1 ***continued***

What is an appropriate measure of economic performance in spaceship earth? It is not the magnitude of material flows, as measured by GNP or the like. On the contrary, it is desirable that the spaceship maintain such flows of material and energy throughput at low levels. Instead, the well-being of the spaceship is best measured by the state – in terms of quality and quantity – of its capital stock, including the state of human minds and bodies.

So, for Boulding, a 'good' state to be in is one in which certain stocks are at high levels – the stock of knowledge, the state of human health, and the stock of capital capable of yielding human satisfaction. Ideally we should aim to make material and energy flows as small as possible to achieve any chosen level of the spaceship's capital stock, maintained over indefinite time.

Boulding is, of course, arguing for a change in our perceptions of the nature of economy–environment interactions, and of what it is that constitutes economic success. He states that

> The shadow of the future spaceship, indeed, is already falling over our spendthrift merriment. Oddly enough, it seems to be in pollution rather than exhaustion, that the problem is first becoming salient. Los Angeles has run out of air, Lake Erie has become a cesspool, the oceans are getting full of lead and DDT, and the atmosphere may become man's major problem in another generation, at the rate at which we are filling it up with junk.

Boulding concludes his paper by considering the extent to which the price mechanism, used in a way to put prices on external diseconomies, can deal with the transition to spaceship earth. He accepts the need for market-based incentive schemes to correct such diseconomies, but argues that these instruments can only deal with a small proportion of the matters which he raises. Boulding concludes:

> The problems which I have been raising in this paper are of larger scale and perhaps much harder to solve . . . One can hope, therefore, that as a succession of mounting crises, especially in pollution, arouse public opinion and mobilize support for the solution of the immediate problems, a learning process will be set in motion which will eventually lead to an appreciation of and perhaps solutions for the larger ones.

Source: Boulding (1966)

To date, the impact of ecological economics on the approach to the natural environment that emerged from mainstream economics has been somewhat limited, and this book will largely reflect that. We will be dealing mainly with mainstream resource and environmental economics, though the next two chapters do address the problem of sustainability. While the theme of sustainability runs through the book, it is not obviously at the forefront in Chapters 4 to 18 which are, mainly, about the mainstream approach. We do, however, at some points in those chapters briefly consider how adopting an ecological economics perspective would affect analysis and policy. In the final chapter of the book, Chapter 19, sustainability returns to the forefront in the context of a discussion of the prospects for promoting it by better economic accounting.

1.3 Fundamental issues in the economic approach to resource and environmental issues

Here we provide a brief anticipatory sketch of four features of the economic approach to resource and environmental issues that will be covered in this book.

1.3.1 Property rights, efficiency and government intervention

We have already stated that a central question in resource and environmental economics concerns allocative efficiency. The role of markets and prices is central to the analysis of this question. As we have noted, a central idea in modern economics is that, given the necessary conditions, markets will bring about efficiency in allocation. Well-defined and enforceable private property rights are one of the necessary conditions. Because property rights do not exist, or are not clearly defined, for many environmental resources, markets fail to allocate those resources efficiently. In such circumstances, price signals fail to reflect true social costs and benefits, and a prima facie case exists for government policy intervention to seek efficiency gains.

Deciding where a case for intervention exists, and what form it should take, is central in all of resource and environmental economics, as we shall see throughout the rest of this book. The foundations for the economic approach to policy analysis are set out in Chapter 4, and the approach is applied in the subsequent chapters. Some environmental problems cross the boundaries of nation states and are properly treated as global problems. In such cases there is no global government with the authority to act on the problem in the same way as the government of a nation state might be expected to deal with a problem within its borders. The special features of international environmental problems are considered in Chapter 9.

1.3.2 The role, and the limits, of valuation, in achieving efficiency

As just observed, many environmental resources – or the services yielded by those resources – do not have well-defined property rights. Clean air is one example of such a resource. Such resources are used, but without being traded through markets, and so will not have market prices. A special case of this general situation is external effects, or externalities. As discussed in Chapter 4, an externality exists where a consumption or production activity has unintended effects on others for which no compensation is paid. Here, the external effect is an untraded – and unpriced – product arising because the victim has no property rights that can be exploited to obtain compensation for the external effect. Sulphur emissions from a coal-burning power station might be an example of this kind of effect.

However, the absence of a price for a resource or an external effect does not mean that it has no value. Clearly, if well-being is affected, there is a value that is either positive or negative depending on whether well-being is increased or decreased. In order to make allocatively efficient decisions, these values need to be estimated in some way. Returning to the power station example, government might wish to impose a tax on sulphur emissions so that the polluters pay for their environmental damage and, hence, reduce the amount of it to the level that goes with allocative efficiency. But this cannot be done unless the proper value can be put on the otherwise unpriced emissions.

There are various ways of doing this – collectively called valuation techniques – which will be explored at some length in Chapter 12. Such techniques are somewhat controversial. There is disagreement between economists over the extent to which the techniques can be expected to produce accurate valuations for unpriced environmental services. These are discussed in Chapter 12. Many non-economists with an interest in how social decisions that affect the environment are made raise rather more fundamental problems about the techniques and their use. Their objection is not, or at least not just, that the techniques may provide the wrong valuations. Rather, they claim that making decisions about environmental services on the basis of monetary valuations of those services is simply the wrong way for society to make such decisions. These objections, and some alternative ways proposed for society to make decisions about the environment, are considered in Chapter 11.

1.3.3 The time dimension of economic decisions

Natural resource stocks can be classified in various ways. A useful first cut is to distinguish between ‘stock’ and ‘flow’ resources. Whereas stock resources, plant and animal populations and mineral deposits, have the characteristic that today’s use has implications for tomorrow’s availability, this is not the case with flow resources. Examples of flow resource are solar radiation, and the power of the wind, of tides and of flowing water. Using more solar radiation today does not itself have any implications for the availability of solar radiation tomorrow. In the case of stock resources, the level of use today does have implications for availability tomorrow.

Within the stock resources category there is an important distinction between ‘renewable’ and ‘non-renewable’ resources. Renewable resources are biotic, plant and animal populations, and have the capacity to grow in size over time, through biological reproduction. Non-renewable resources are abiotic, stocks of minerals, and do not have that capacity to grow over time. What are here called non-renewable resources are sometimes referred to as ‘exhaustible’, or ‘depletable’, resources. This is because there is no positive constant rate of use that can be sustained indefinitely – eventually the resource stock must be exhausted. This is not actually a useful terminology. Renewable resources are exhaustible if harvested for too long at a rate exceeding their regeneration capacities.

From an economic perspective, stock resources are assets yielding flows of environmental services over time. In considering the efficiency and optimality of their use, we must take account not only of use at a point in time but also of the pattern of use over time. Efficiency and optimality have, that is, an intertemporal, or dynamic, dimension, as well as an intratemporal, or static, dimension. Chapter 11 sets out the basics of intertemporal welfare economics. In thinking about the intertemporal dimension of the use of environmental resources, attention must be given to the productiveness of the capital that is accumulated as a result of saving and investment. If, by means of saving and investment, consumption is deferred to a later period, the increment to future consumption that follows from such investment will generally exceed the initial consumption quantity foregone. The size of the pay-off to deferred consumption is reflected in the rate of return to investment.

Environmental resource stocks similarly have rates of return associated with their deferred use. The relations between rates of return to capital as normally understood in economics and the rates of return on environmental assets must be taken into account in trying to identify efficient and optimal paths of environmental resource use over time. The arising theory of the efficient and optimal use of natural and environmental resources over time is examined in Chapters 14, 15, 17 and 18, and is drawn on in Chapter 19. As discussed in Chapter 16, many pollution problems also have an intertemporal dimension, and it turns out that the analysis developed for thinking about the intertemporal problems of resource use can be used to analyse those problems.

1.3.4 Substitutability and irreversibility

Substitutability and irreversibility are important, and related, issues in thinking about policy in relation to the natural environment. If the depletion of a resource stock is irreversible, and there is no close substitute for the services that it provides, then clearly the rate at which the resource is depleted has major implications for sustainability. To the extent that depletion is not irreversible and close substitutes exist, there is less cause for concern about the rate at which the resource is used.

There are two main dimensions to substitutability issues. First, there is the question of the extent to which one natural resource can be replaced by another. Can, for example, solar power substitute for the fossil fuels on a large scale? This is, as we shall see, an especially important question given that the combustion of fossil fuels not only involves the depletion of non-renewable resources, but also is a source of some major environmental pollution problems, such as the so-called greenhouse effect which entails the prospect of global climate change, as discussed in Chapter 9.

Second, there is the question of the degree to which an environmental resource can be replaced by other inputs, especially the human-made capital resulting from saving and investment. As we shall see, in Chapters 3, 14 and 19, this question is of particular significance when we address questions concerning long-run economy–environment interactions, and the problem of sustainability.

Human-made capital is sometimes referred to as reproducible capital, identifying an important difference between stocks of it and stocks of non-renewable resources. The latter are not reproducible, and their exploitation is irreversible in a way that the use of human-made capital is not. We shall discuss this further in the next chapter, and some arising implications in later chapters, especially 14 and 19. With renewable resource stocks, depletion is reversible to the extent that harvesting is at rates that allow regeneration. Some of the implications are discussed in Chapter 17. Some pollution problems may involve irreversible effects, and the extinction of a species of plant or animal is certainly irreversible.

Some assemblages of environmental resources are of interest for the amenity services, recreation and aesthetic enjoyment that they provide, as well as for their potential use as inputs to production. A wilderness area, for example, could be conserved as a national park or developed for mining. Some would also argue that there are no close substitutes for the services of wilderness. A decision to develop such an area would be effectively irreversible, whereas a decision to conserve would be reversible. We show in Chapter 13 that under plausible conditions this asymmetry implies a stronger preference for non-development than would be the case where all decisions are reversible, and that this is strengthened when it is recognised that the future is not known with certainty. Imperfect knowledge of the future is, of course, the general condition, but it is especially relevant to decision making about many environmental problems, and has implications for how we think about making such decisions.

1.4 Reader's guide

We have already noted in which chapters various topics are covered. Now we will briefly set out the structure of this text, and explain the motivation for that structure.

In Part I we deal with 'Foundations' of two kinds. First, in Chapter 2, we explain why many people think that there is a sustainability problem. We consider the interdependence of the economy and the environment, look at the current state of human development, and at some views on future prospects. Second, in the next two chapters, we work through the conceptual basis and the analytical tools with which economists approach environmental problems. Chapter 3 looks at the ethical basis for policy analysis in economics. Chapter 4 is about welfare economics and markets – what they achieve when they work properly, why they do not always work properly, and what can be done about it when they do not work properly.

Throughout the book we have put as much of the mathematics as is possible in appendices, of which extensive use is made. These appendices will be found on the companion website for this book: www.pearsoned.co.uk/perman. Readers who have learned the essentials of the calculus of constrained optimisation will have no problems with the mathematics used in the appendices in Part I. Appendix 3.1 provides a brief account of the mathematics of constrained optimisation. The arguments of Part I can be followed without using the mathematics in the appendices, but readers who work through them will obtain a deeper understanding of the arguments and their foundations.

Part II is about 'Environmental pollution'. It turns out that much, but not all, of what economists have to say about pollution problems relates to the question of intratemporal allocative efficiency and does not essentially involve a time dimension. The static analysis of pollution problems is the focus of Part II. Static, as opposed to dynamic, analysis follows naturally from the material covered most intensively in Chapter 4, and, subject to an exception to be noted shortly, the mathematics used in the appendices in Part II is of the same kind as used in the appendices in Part I.

Chapter 5 considers the setting of targets for pollution control, and Chapter 6 looks at the analysis of the policy instruments that could be used to meet those targets. In these chapters it is assumed that the government agency responsible for pollution control has complete information about all aspects of the pollution problem to be addressed. This is a patently unrealistic assumption, and Chapter 7 examines the consequences of its relaxation. The analysis in these three chapters is partial, analysing the control of a particular pollutant as if it were the only such problem, and as if what were done about it had no implications for the rest of the economy. Chapter 8, in contrast, takes an approach which looks at the economy as a whole, using input–output analysis and introducing applied general equilibrium modelling. This chapter includes an appendix that provides a brief review of the matrix algebra which facilitates the understanding and application of these methods. Part II finishes with Chapters 9 and 10, which deal, respectively, with environmental problems that cross national frontiers, and with how thinking about international trade is affected by the existence of environmental problems.

Part III has the title 'Project appraisal'. Its focus is on the rationale for, and application of, the methods and techniques that economists have developed for evaluating whether going ahead with some discrete investment project, or policy innovation, is in the public interest. Of particular concern here, of course, are projects and policies with environmental impacts. Also, the focus is on projects and policies which have consequences stretching out over time. Chapter 11 deals with the principles of intertemporal welfare economics and their application in cost–benefit analysis. Chapter 11 also looks at some alternative methods for project appraisal that have been advocated, especially by those who have ethical objections to the use of cost–benefit analysis where the natural environment is involved. A necessary input to a cost–benefit analysis of a project with effects on the natural environment is a monetary evaluation of those effects. The methods that economists have devised for monetary evaluation of non-marketed environmental services are explained in Chapter 12. Chapter 13 looks at the implications for project appraisal of recognition of the facts that when looking at projects with environmental impacts, we are often dealing with impacts that are irreversible, and always considering future effects about which our knowledge is incomplete.

In Part III the arguments and analysis are developed mainly in the context of the recreation and amenity services that the natural environment provides, though they are, of course, also relevant to the problem of environmental pollution, the focus of Part II. In Part IV we turn to a focus on the issues associated with the extraction of natural resources from the environment for use as inputs to production. The problems that have most interested economists here are essentially dynamic in nature, that is, are problems in intertemporal allocation. In addressing these problems, economists typically use the mathematics of 'optimal control'. We have minimised the explicit use of this mathematics in the body of the text, but we do make extensive use of it in the appendices in Part IV. For readers not familiar with this sort of mathematics, Appendix 14.1 provides a brief account of it, treating it as an extension of the ideas involved in ordinary constrained optimisation developed in Appendix 3.1.

Chapter 14 introduces the application of the basic ideas about intertemporal optimality and efficiency, developed in Chapter 11, to the question of natural resource extraction. Chapter 15 looks specifically at the extraction of non-renewable resources, that is, stocks of minerals and fossil fuels. The case of renewable resources – populations of plants and animals harvested for use in production and consumption – is dealt with in Chapter 17. Trees are plants with some special characteristics, and Chapter 18 reviews the major elements of forestry economics. Many important pollution problems have the characteristic that the pollutant involved accumulates in the environment as a stock, which may decay naturally over time. Analysis of such pollution problems has much in common with the analysis of natural resource extraction, and is dealt with in Chapter 16. Finally in Part IV we return to the sustainability issue. Chapter 19 is about modifying standard accounting procedures so as to have economic performance indicators reflect environmental impacts, and particularly so as to measure sustainable national income.

Summary

There is not a single methodology used by all economists working on matters related to natural resources and the environment. Ecological economists have argued the need to work towards a more holistic discipline that would integrate natural-scientific and economic paradigms. Some ecological economists argue further that the sustainability problem requires nothing less than a fundamental change in social values, as well as a scientific reorientation. While some movement has been made in the direction of interdisciplinary cooperation, most analysis is still some way from having achieved integration. At the other end of a spectrum of methodologies are economists who see no need to go beyond the application of neoclassical techniques to environmental problems, and stress the importance of constructing a more complete set of quasi-market incentives to induce efficient behaviour. Such economists would reject the idea that existing social values need to be questioned, and many have great faith in the ability of continuing technical progress to ameliorate problems of resource scarcity and promote sustainability. Ecological economists tend to be more sceptical about the extent to which technical progress can overcome the problems that follow from the interdependence of economic and environmental systems.

However, there is a lot of common ground between economists working in the area, and it is this that we mainly focus upon in this text. Nobody who has seriously studied the issues believes that the economy's relationship to the natural environment can be left entirely to market forces. Hardly anybody now argues that market-like incentives have no role to play in that relationship. In terms of policy, the arguments are about how much governments need to do, and the relative effectiveness of different kinds of policy instruments. Our aim in this book is to work through the economic analysis relevant to these kinds of questions, and to provide information on the resource and environmental problems that they arise from. We begin, in the next chapter, by discussing the general interdependence of the economic and environmental systems, and the concerns about sustainability that this has given rise to.

Further reading

As all save one of the topics and issues discussed in this chapter will be dealt with more comprehensively in subsequent chapters, we shall not make any suggestions for further reading here other than for that one topic – the history of economics. Blaug (1985), *Economic Theory in Retrospect*, is essential reading for anybody who wants to study the history of economic ideas in detail. For those who do not require a comprehensive treatment, useful alternatives are Barber (1967) and Heilbronner (1991). Crocker (1999) is a short overview of the history of environmental and resource economics, providing references to seminal contributions.

The leading specialist journals, in order of date of first issue, are: *Land Economics*, *Journal of Environmental Economics and Management*, *Ecological Economics*, *Environmental and Resource Economics*, *Environment and Development Economics*. The first issue of *Ecological Economics*, February 1989, contains several articles on the nature of ecological economics. The May 2000 issue, Vol. 39, number 3, of the *Journal of Environmental Economics and Management* marks the journal's 25th anniversary and contains articles reviewing the major developments in environmental and resource economics over its lifetime.

The *Journal of Environmental Economics and Management* is run by the Association of Environmental and Resource Economists, whose website at www.aere.org has useful information and links. The equivalent European association is the European Association of Environmental and Resource Economists – www.vwl.unimannheim.de/conrad/eaere/ – which runs the journal *Environmental and Resource Economics*. The address for the website of the International Society for Ecological Economics is www.ecoeco.org/.

CHAPTER 2 The origins of the sustainability problem

The global challenge can be simply stated: To reach sustainability, humanity must increase the consumption levels of the world's poor, while at the same time reducing humanity's ecological footprint

Meadows *et al.* (2005), p. xv

Learning objectives

In this chapter you will

- learn how economic activity depends upon and affects the natural environment
- be introduced to some basic material from the environmental sciences
- learn about the proximate drivers of the economy's impact on the environment – population, affluence and technology
- review the current state of human economic development
- consider the argument that the environment sets limits to economic growth
- learn about the emergence of the idea of sustainable development

Introduction

We inhabit a world in which the human population has risen dramatically over the past century and may almost double during the next. The material demands being made by the average individual have been increasing rapidly, though many human beings now alive are desperately poor. Since the 1950s and 1960s economic growth has been generally seen as *the* solution to the problem of poverty. Without economic growth, poverty alleviation involves redistribution from the better-off to the poor, which encounters resistance from the better-off. In any case, there may be so many poor in relation to the size of the better-off group that the redistributive solution to the problem of poverty is simply impossible – the cake is not big enough to provide for all, however thinly the slices are cut. Economic growth increases the size of the cake. With enough of it, it may be possible to give everybody at least a decent slice, without having to reduce the size of the larger slices.

However, the world's resource base is limited, and contains a complex, and interrelated, set of ecosystems that are currently exhibiting signs of fragility. It is increasingly questioned whether the global economic system can continue to grow without undermining the natural systems which are its ultimate foundation.

This set of issues we call 'the sustainability problem' – how to alleviate poverty in ways that do not affect the natural environment such that future economic prospects suffer. In this chapter we set out the basis for the belief that such a problem exists.

This chapter is organised as follows. We first look at the interdependence of the economy and the environment, and give a brief overview of some environmental science basics that are relevant to this. In the second section the proximate drivers of the economy's impact on the environment are considered. The third section of the chapter presents data on the current state of human development in relation to the problems of poverty and inequality. In this section we note the attachment of economists to economic growth as the solution to the poverty problem. In the next section we consider limits to growth. The chapter ends by looking at the emergence in the 1980s of the idea of sustainable development – growth that does not damage the environment – and progress towards its realisation.

2.1 Economy–environment interdependence

Economic activity takes place within, and is part of, the system which is the earth and its atmosphere. This system we call 'the natural environment', or more briefly 'the environment'. This system itself has an environment, which is the rest of the universe. Figure 2.1 is a schematic representation of the two-way relationships between, the interdependence of, the economy and the environment.[1]

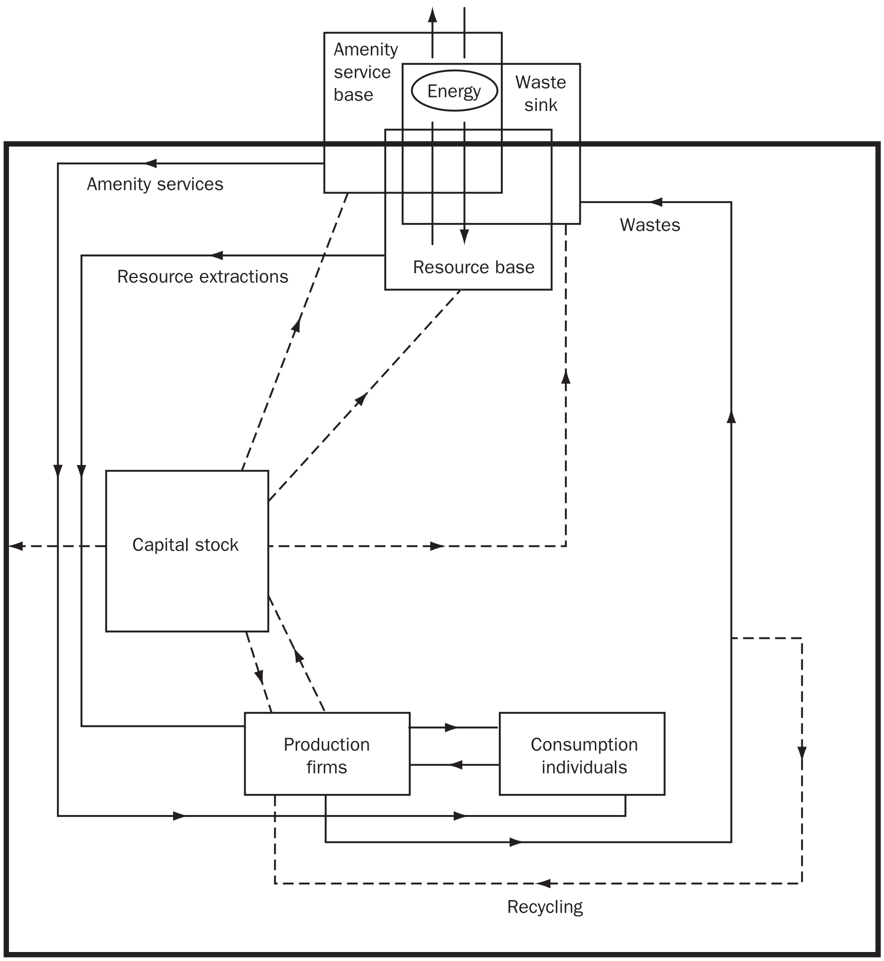


Figure 2.1 Economic activity in the natural environment

[1] Figure 2.1 is taken from Common (1995), where economy–environment interdependence is discussed at greater length than here. References to works which deal more fully, and rigorously, with the natural science matters briefly reviewed here are provided in the Further Reading section at the end of the chapter.

The outer heavy black lined box represents the environment, which is a thermodynamically closed system, in that it exchanges energy but not matter with its environment. The environment receives inputs of solar radiation. Some of that radiation is absorbed and drives environmental processes. Some is reflected back into space. This is represented by the arrows crossing the heavy black line at the top of the figure. Matter does not cross the heavy black line. The balance between energy absorption and reflection determines the way the global climate system functions. The energy in and out arrows are shown passing through three boxes, which represent three of the functions that the environment performs in relation to economic activity. The fourth function, represented by the heavy black lined box itself, is the provision of the life-support services and those services which hold the whole functioning system together. Note that the three boxes intersect one with another and that the heavy black line passes through them. This is to indicate that the four functions interact with one another, as discussed below.

Figure 2.1 shows economic activity located within the environment and involving production and consumption, both of which draw upon environmental services, as shown by the solid lines inside the heavy lined box. Not all of production is consumed. Some of the output from production is added to the human-made, reproducible, capital stock, the services of which are used, together with labour services, in production. Figure 2.1 shows production using a third type of input, resources extracted from the environment. Production gives rise to wastes inserted into the environment. So does consumption. Consumption also uses directly a flow of amenity services from the environment to individuals without the intermediation of productive activity.

We now discuss these four environmental functions, and the interactions between them, in more detail.

2.1.1 The services that the environment provides

As noted in the previous chapter, natural resources used in production are of several types. One distinguishing characteristic is whether the resource exists as a stock or a flow. The difference lies in whether the level of current use affects future availability. In the case of flow resources there is no link between current use and future availability. The prime example of a flow resource is solar radiation – if a roof has a solar water heater on it, the amount of water heating done today has no implications for the amount that can be done tomorrow. Wave and wind power are also flow resources. Stock resources are defined by the fact that the level of current use does affect future availability.

Within the class of stock resources, a second standard distinction concerns the nature of the link between current use and future availability. Renewable resources are biotic populations – flora and fauna. Non-renewable resources are minerals, including the fossil fuels. In the former case, the stock existing at a point in time has the potential to grow by means of natural reproduction. If in any period use of the resource is less than natural growth, stock size grows. If use, or harvest, is always the same as natural growth, the resource can be used indefinitely. Such a harvest rate is often referred to as a ‘sustainable yield’. Harvest rates in excess of sustainable yield imply declining stock size. For non-renewable resources there is no natural reproduction, except on geological timescales. Consequently, more use now necessarily implies less future use.

Within the class of non-renewables the distinction between fossil fuels and the other minerals is important. First, the use of fossil fuels is pervasive in industrial economies, and could be said to be one of their essential distinguishing characteristics. Second, fossil fuel combustion is an irreversible process in that there is no way in which the input fuel can be even partially recovered after combustion. In so far as coal, oil and gas are used to produce heat, rather than as inputs to chemical processes, they cannot be recycled. Minerals used as inputs to production can be recycled. This means that whereas in the case of minerals there exists the possibility of delaying, for a given use rate, the date of exhaustion of a given initial stock, in the case of fossil fuels there does not. Third, fossil fuel combustion is a major source of a number of waste emissions, especially into the atmosphere. One such emission is carbon dioxide, CO_2, the most important of the greenhouse gases which are driving climate change.

Many of the activities involved in production and consumption give rise to waste products, or residuals, to be discharged into the natural environment. Indeed, as we shall see when we discuss the materials balance principle, such insertions into the environment are the necessary corollary of the extraction of material resources from it. In economics, questions relating to the consequences of waste discharge into the environment are generally discussed under the heading of 'pollution'. To the extent, and only to the extent, that waste discharge gives rise to problems perceived by humans, economists say that there is a pollution problem. Pollution problems can be conceptualised in two ways. One, which finds favour with economists, sees pollution as a stock of material resident in the natural environment. The other, which finds favour more with ecologists, sees pollution as a flow which affects the natural environment.

In the former case, pollution is treated in the same way as a stock resource, save that the stock has negative value. Residual flows into the environment add to the stock; natural decay processes subtract from it. We will look at pollution modelled this way in Chapter 16. The flow model treats the environment as having an 'assimilative capacity', defined in terms of a rate of residual flow. Pollution is then the result of a residual flow rate in excess of assimilative capacity. There is no pollution if the residual flow rate is equal to, or less than, assimilative capacity. If the residual flow rate is persistently in excess of assimilative capacity, the latter declines over time, and may eventually go to zero.

In Figure 2.1 amenity services flow directly from the environment to individuals. The biosphere provides humans with recreational facilities and other sources of pleasure and stimulation. Swimming from an ocean beach does not require productive activity to transform an environmental resource into a source of human satisfaction, for example. Wilderness recreation is defined by the absence of other human activity. Some people like simply lying out of doors in sunshine. The role of the natural environment in regard to amenity services can be appreciated by imagining its absence, as would be the case for the occupants of a space vehicle. In many cases the flow to individuals of amenity services does not directly involve any consumptive material flow. Wilderness recreation, for example, is not primarily about consuming resources in the wilderness area, though it may involve this in the use of wood for fires, the capture of game for food and so on. A day on the beach does not involve any consumption of the beach in the way that the use of oil involves its consumption. This is not to say that flows of amenity services never impact physically on the natural environment. Excessive use of a beach area can lead to changes in its character, as with the erosion of sand dunes following vegetation loss caused by human visitation.

The fourth environmental function, shown in Figure 2.1 as the heavy box, is difficult to represent in a simple and concise way. Over and above serving as resource base, waste sink and amenity base, the biosphere currently provides the basic life-support functions for humans. While the range of environmental conditions that humans are biologically equipped to cope with is greater than for most other species, there are limits to the tolerable. We have, for example, quite specific requirements in terms of breathable air. The range of temperatures that we can exist in is wide in relation to conditions on earth, but narrow in relation to the range on other planets in the solar system. Humans have minimum requirements for water input. And so on. The environment functions now in such a manner that humans can exist in it. An example will illustrate what is involved.

Consider solar radiation. It is one element of the resource base, and for some people sunbathing is an environmental amenity service. In fact, solar radiation as it arrives at the earth's atmosphere is harmful to humans. There it includes the ultraviolet wavelength UV-B, which causes skin cancer, adversely affects the immune system and can cause eye cataracts. UV-B radiation affects other living things as well. Very small organisms are likely to be particularly affected, as UV-B can penetrate only a few layers of cells. This could be a serious problem for marine systems, where the base of the food chain consists of very small organisms living in the surface layers of the ocean, which UV-B reaches. UV-B radiation also affects photosynthesis in green plants adversely.

Solar radiation arriving at the surface of the earth has much less UV-B than it does arriving at the atmosphere. Ozone in the stratosphere absorbs UV-B, performing a life-support function by filtering solar radiation. In the absence of stratospheric ozone, it is questionable whether human life could exist. Currently, stratospheric ozone is being depleted by the release into the atmosphere of chlorofluorocarbons (CFCs), compounds which exist only by virtue of human economic activity. They have been in use since the 1940s. Their ozone-depleting properties were recognised in the 1980s, and, as discussed in Chapter 9, policy to reduce this form of pollution is now in place.

The interdependencies between economic activity and the environment are pervasive and complex. The complexity is increased by the existence of processes in the environment that mean that the four classes of environmental services each interact one with another. In Figure 2.1 this is indicated by having the three boxes intersect one with another, and jointly with the heavy black line representing the life-support function. What is involved can be illustrated with the following example.

Consider a river estuary. It serves as resource base for the local economy in that a commercial fishery operates in it. It serves as waste sink in that urban sewage is discharged into it. It serves as the source of amenity services, being used for recreational purposes such as swimming and boating. It contributes to life-support functions in so far as it is a breeding ground for marine species which are not commercially exploited, but which play a role in the operation of the marine ecosystem. At rates of sewage discharge equal to or below the assimilative capacity of the estuary, all four functions can coexist. If, however, the rate of sewage discharge exceeds assimilative capacity, not only does a pollution problem emerge, but the other estuarine functions are impaired. Pollution will interfere with the reproductive capacity of the commercially exploited fish stocks, and may lead to the closure of the fishery. This does not necessarily mean its biological extinction. The fishery may be closed on the grounds of danger to public health. Pollution will reduce the capacity of the estuary to support recreational activity, and in some respects, such as swimming, may drive it to zero. Pollution will also impact on the non-commercial marine species, and may lead to their extinction, with implications for marine ecosystem function.

An example at the global level of the interconnections between the environmental services arising from interacting environmental processes affected by economic activity is provided by the problem of global climate change, which is discussed below and, at greater length, in Chapter 9.

2.1.2 Substituting for environmental services

One feature of Figure 2.1 remains to be considered. We have so far discussed the solid lines. There are also some dashed lines. These represent possibilities of substitutions for environmental services.

Consider first recycling. This involves interception of the waste stream prior to its reaching the natural environment, and the return of some part of it to production. Recycling substitutes for environmental functions in two ways. First, it reduces the demands made upon the waste sink function. Second, it reduces the demands made upon the resource base function, in so far as recycled materials are substituted for extractions from the environment.

Also shown in Figure 2.1 are four dashed lines from the box for capital running to the three boxes and the heavy black line representing environmental functions. These lines are to represent possibilities for substituting the services of reproducible capital for environmental services. Some economists think of the environment in terms of assets that provide flows of services, and call the collectivity of environmental assets 'natural capital'. In that terminology, the dashed lines refer to possibilities for substituting reproducible capital services for natural capital services.

In relation to the waste sink function consider again, as an example, the discharge of sewage into a river estuary. Various levels of treatment of the sewage prior to its discharge into the river are possible. According to the level of treatment, the demand made upon the assimilative capacity of the estuary is reduced for a given level of sewage. Capital in the form of a sewage treatment plant substitutes for the natural environmental function of waste sink to an extent dependent on the level of treatment that the plant provides.

An example from the field of energy conservation illustrates the substitution of capital for resource base functions. For a given level of human comfort, the energy use of a house can be reduced by the installation of insulation and control systems. These add to that part of the total stock of capital equipment which is the house and all of its fittings, and thus to the total capital stock. Note, however, that the insulation and control systems are themselves material structures, the production of which involves extractions, including energy, from the environment. Similar fuel-saving substitution possibilities exist in productive activities.

Consider next some examples in the context of amenity services. An individual who likes swimming can do this in a river or lake, or from an ocean beach, or in a manufactured swimming pool. The experiences involved are not identical, but they are close substitutes in some dimensions. Similarly, it is not now necessary to actually go into a natural environment to derive pleasure from seeing it. The capital equipment in the entertainment industry means that it is possible to see wild flora and fauna without leaving an urban environment. Apparently it is envisaged that computer technology will, via virtual reality devices, make it possible to experience many of the sensations involved in being in a natural environment without actually being in it.

It appears that it is in the context of the life-support function that many scientists regard the substitution possibilities as most limited. However, from a purely technical point of view, it is not clear that this is the case. Artificial environments capable of supporting human life have already been created, and in the form of space vehicles and associated equipment have already enabled humans to live outside the biosphere, albeit in small numbers and for limited periods. It would apparently be possible, if expensive, to create conditions capable of sustaining human life on the moon, given some suitable energy source. However, the quantity of human life that could be sustained in the absence of natural life-support functions would appear to be quite small. It is not that those functions are absolutely irreplaceable, but that they are irreplaceable on the scale that they operate. A second point concerns the quality of life. One might reasonably take the view that while human life on an otherwise biologically dead earth is feasible, it would not be in the least desirable.

The possibilities for substituting for the services of natural capital have been discussed in terms of capital equipment. Capital is accumulated when output from current production is not used for current consumption. Current production is not solely of material structures, and reproducible capital does not only comprise equipment – machines, buildings, roads and so on. 'Human capital' is increased when current production is used to add to the stock of knowledge, and is what forms the basis for technical change. However, while the accumulation of human capital is clearly of great importance in regard to environmental problems, in order for technical change to impact on economic activity it generally requires embodiment in new equipment. Knowledge that could reduce the demands made upon environmental functions does not actually do so until it is incorporated into equipment that substitutes for environmental functions.

Capital for environmental service substitution is not the only form of substitution that is relevant to economy–environment interconnections. In Figure 2.1 flows between the economy and the environment are shown as single lines. Of course, each single line represents what is in fact a whole range of different flows. With respect to each of the aggregate flows shown in Figure 2.1, substitutions as between components of the flow are possible and affect the demands made upon environmental services. The implications of any given substitution may extend beyond the environmental function directly affected. For example, a switch from fossil fuel use to hydro-electric power reduces fossil fuel depletion and waste generation in fossil fuel combustion, and also impacts on the amenity service flow in so far as a natural recreation area is flooded.

2.1.3 Some environmental science

We now briefly review some elements of the environmental sciences which are important to an understanding of the implications of economy–environment interdependence. The review is necessarily very selective; references to useful supplementary reading are provided at the end of the chapter.

2.1.3.1 Thermodynamics

Thermodynamics is the science of energy. Energy is the potential to do work or supply heat. It is a characteristic of things, rather than a thing itself. Work is involved when matter is changed in structure, in physical or chemical nature, or in location. In thermodynamics it is necessary to be clear about the nature of the system under consideration. An 'open' system is one which exchanges energy and matter with its environment. An individual organism – a human being for example – is an open system. A 'closed' system exchanges energy, but not matter, with its environment. Planet earth and its atmosphere are a closed system. An 'isolated' system exchanges neither energy nor matter with its environment. Apart from the entire universe, an isolated system is an ideal, an abstraction.

The first law of thermodynamics says that energy can neither be created nor destroyed – it can only be converted from one form to another. Many of those who are concerned about the environment want to encourage people to go in for 'energy conservation'. But, the first law says that there is always 100% energy conservation whatever people do. There is no real contradiction here, just an imprecise use of language on the part of those seeking to promote 'energy conservation'. What they actually want to encourage is people doing the things that they do now but in ways that require less heat and/or less work, and therefore less energy conversion.

The second law of thermodynamics is also known as 'the entropy law'. It says that heat flows spontaneously from a hotter to a colder body, and that heat cannot be transformed into work with 100% efficiency. It follows that all conversions of energy from one form to another are less than 100% efficient. This appears to contradict the first law, but does not. The point is that not all of the energy of some store, such as a fossil fuel, is available for conversion. Energy stores vary in the proportion of their energy that is available for conversion. 'Entropy' is a measure of unavailable energy. All energy conversions increase the entropy of an isolated system. All energy conversions are irreversible, since the fact that the conversion is less than 100% efficient means that the work required to restore the original state is not available in the new state. Fossil fuel combustion is irreversible, and of itself implies an increase in the entropy of the system which is the environment in which economic activity takes place. However, that environment is a closed, not an isolated, system, and is continually receiving energy inputs from its environment, in the form of solar radiation. This is what makes life possible.

Thermodynamics is difficult for non-specialists to understand. Even within physics it has a history involving controversy, and disagreements persist, as will be noted below. There exist some popular myths about thermodynamics and its implications. It is, for example, often said that entropy always increases. This is true only for an isolated system. Classical thermodynamics involved the study of equilibrium systems, but the systems directly relevant to economic activity are open and closed systems which are far from equilibrium. Such systems receive energy from their environment. As noted above, a living organism is an open system, which is far from equilibrium. Some energy input is necessary for it to maintain its structure and not become disordered – in other words, dead.

The relevance of thermodynamics to the origins of the problem of sustainability is clear. The economist who did most to try to make his colleagues aware of the laws of thermodynamics and their implications, Nicholas Georgescu-Roegen (who started academic life as a physicist), described the second law as the 'taproot of economic scarcity' (Georgescu-Roegen, 1979). His point was, to put it graphically, that if energy conversion processes were 100% efficient, one lump of coal would last for ever.

Material transformations involve work, and thus require energy. Given a fixed rate of receipt of solar energy, there is an upper limit to the amount of work that can be done on the basis of it. For most of human history, human numbers and material consumption levels were subject to this constraint. The exploitation of fossil fuels removes this constraint. The fossil fuels are accumulated past solar energy receipts, initially transformed into living tissue, and stored by geological processes. Given this origin, there is necessarily a finite amount of the fossil fuels in existence. It follows that in the absence of an abundant substitute energy source with similar qualities to the fossil fuels, such as nuclear fusion, there would eventually be a reversion to the energetic situation of the pre-industrial phase of human history, which involved total reliance on solar radiation and other flow sources of energy. Of course, the technology deployed in such a situation would be different from that available in the pre-industrial phase. It is now possible, for example, to use solar energy to generate electricity.

2.1.3.1.1 Recycling

The laws of thermodynamics are generally taken to mean that, given enough available energy, all transformations of matter are possible, at least in principle. On the basis of that understanding it has generally been further understood that, at least in principle, complete material recycling is possible. On this basis, given the energy, there is no necessity that shortage of minerals constrain economic activity. Past extractions could be recovered by recycling. It is in this sense that the second law of thermodynamics is the ultimate source of scarcity. Given available energy, there need be no scarcity of minerals. This is what drives the interest in nuclear power, and especially nuclear fusion, which might offer the prospect of a clean and effectively infinite energy resource.

Nicholas Georgescu-Roegen, noted above as the economist who introduced the idea of the second law as the ultimate basis for economic scarcity, subsequently attacked the view just sketched as 'the energetic dogma', and insisted that 'matter matters' as well (Georgescu-Roegen, 1979). He argued that even given enough energy, the complete recycling of matter is, in principle, impossible. This has been dubbed 'the fourth law of thermodynamics' and its validity has been denied: e.g. 'complete recycling is physically possible if a sufficient amount of energy is available' (Biancardi *et al*., 1993). The basis for this denial is that the fourth law would be inconsistent with the second. This disagreement over what is a very basic scientific issue is interesting for two reasons. First, if qualified scientists can disagree over so fundamental a point, then it is clear that many issues relevant to sustainability involve uncertainty. Secondly, both sides to this dispute would agree that, as a practical matter, complete recycling is impossible however much energy is available. Thus, the statement above rebutting the fourth law is immediately followed by: 'The problem is that such expenditure of energy would involve a tremendous increase in the entropy of the environment, which would not be sustainable for the biosphere' (Biancardi *et al*., 1993). Neither party to the dispute is suggesting that policy should be determined on the basis of an understanding that matter can actually be completely recycled.

2.1.3.2 The materials balance principle

'The materials balance principle' is the term that economists tend to use to refer to the law of conservation of mass, which states that matter can neither be created nor destroyed. An early exposition of the principle as it applies to economic activity is found in Kneese *et al*. (1970). As far as economics goes, the most fundamental implication of the materials balance principle is that economic activity essentially involves transforming matter extracted from the environment. Economic activity cannot, in a material sense, create anything. It does, of course, involve transforming material extracted from the environment so that it is more valuable to humans. But, another implication is that all of the material extracted from the environment must, eventually, be returned to it, albeit in a transformed state. The 'eventually' is necessary because some of the extracted material stays in the economy for a long time – in buildings, roads, machinery and so on.

Figure 2.2 shows the physical relationships implied by the materials balance principle. It abstracts from the lags in the circular flow of matter due to capital accumulation in the economy. It amplifies the picture of material extractions from and insertions into the environment provided in Figure 2.1. Primary inputs (ores, liquids and gases) are taken from the environment and converted into useful products (basic fuel, food and raw materials) by 'environmental' firms. These outputs become inputs into subsequent production processes (shown as a product flow to non-environmental firms) or to households directly. Households also receive final products from the non-environmental firms sector.

The materials balance principle states an identity between the mass of materials flow from the environment (flow A) and the mass of residual material discharge flows to the environment (flows $B + C + D$). So, in terms of mass, we have $A \equiv B + C + D$. In fact several identities are implied by Figure 2.2. Each of the four sectors shown by rectangular boxes receives an equal mass of inputs to the mass of its outputs. So we have the following four identities:

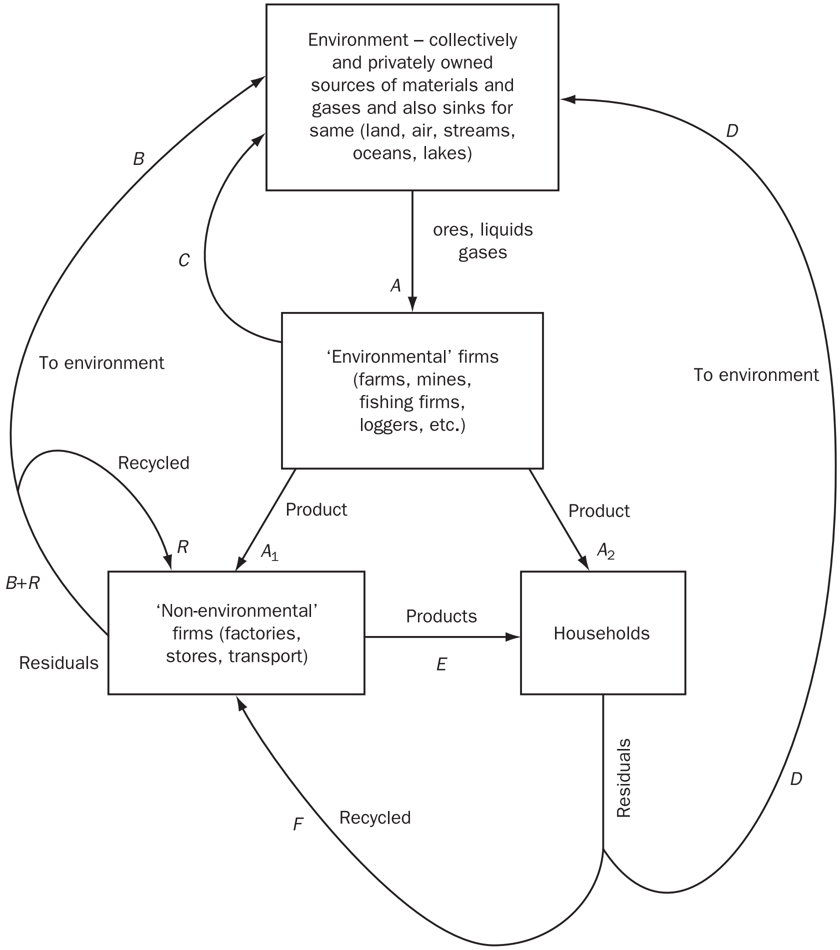


Figure 2.2 A materials balance model of economy–environment interactions
Source: Adapted from Herfindahl & Kneese (1974)

The environment:	$A \equiv B + C + D$ as above
Environmental firms:	$A \equiv A_1 + A_2 + C$
Non-environmental firms:	$B + R + E \equiv R + A_1 + F$
Households:	$A_2 + E \equiv D + F$

Several insights can be derived from this model. First, in a materially closed economy in which no net stock accumulation takes place (that is, physical assets do not change in magnitude) the mass of residuals into the environment ($B + C + D$) must be equal to the mass of fuels, foods and raw materials extracted from the environment and oxygen taken from the atmosphere (flow A). Secondly, the treatment of residuals from economic activity does not reduce their mass, although it alters their form. Nevertheless, while waste treatment does not 'get rid of' residuals, waste management can be useful by transforming residuals to a more benign form (or by changing their location).

Thirdly, the extent of recycling is important. To see how, look again at the identity $B + R + E \equiv R + A_1 + F$. For any fixed magnitude of final output, E, if the amount of recycling of household residuals, F, can be increased, then the quantity of inputs into final production, A_1, can be decreased. This in turn implies that less primary extraction of environmental resources, A, need take place. So the total amount of material throughput in the system (the magnitude A) can be decreased for any given level of production and consumption if the efficiency of materials utilisation is increased through recycling processes.

2.1.3.2.1 Production function specification

In most of microeconomics, production is taken to involve inputs of capital and labour. For the ith firm, the production function is written as

$$Q_i = f_i(L_i, K_i) \tag{2.1}$$

where Q represents output, L labour input and K capital input. According to the materials balance principle, this cannot be an adequate general representation of what production involves. If Q_i has some material embodiment, then there must be some material input to production – matter cannot be created.

If we let R represent some natural resource extracted from the environment, then the production function could be written as:

$$Q_i = f_i(L_i, K_i, R_i) \tag{2.2}$$

Production functions with these arguments are widely used in the resource economics literature. In contrast, the environmental economics literature tends to stress insertions into the environment – wastes arising in production and consumption wastes – and often uses a production function of the form

$$Q_i = f_i(L_i, K_i, M_i) \tag{2.3}$$

where M_i is the flow of waste arising from the ith firm's activity. Equation 2.3 may appear strange at first sight as it treats waste flows as an input into production. However, this is a reasonable way of proceeding given that reductions in wastes will mean reductions in output for given levels of the other inputs, as other inputs have to be diverted to the task of reducing wastes.

A more general version of equation 2.3 is given by

$$Q_i = f_i\left(L_i, K_i, M_i, A\left[\sum_i M_i\right]\right) \tag{2.4}$$

in which A denotes the ambient concentration level of some pollutant, which depends on the total of waste emissions across all firms. Thus, equation 2.4 recognises that ambient pollution can affect production possibilities.

However, as it stands, equation 2.4 conflicts with the materials balance principle. Now, matter in the form of waste is being created by economic activity alone, which is not possible. A synthesis of resource and environmental economics production functions is desirable, which recognises that material inputs (in the form of environmental resources) enter the production function and material outputs (in the form of waste as well as output) emanate from production. This yields a production function such as

$$Q_i = f_i\left(L_i, K_i, R_i, M_i[R_i], A\left[\sum_i M_i\right]\right) \tag{2.5}$$

Where some modelling procedure requires the use of a production function, the use of a form such as equation 2.5 has the attractive property of recognising that, in general, production must have a material base, and that waste emissions necessarily arise from that base. It is consistent, that is, with one of the fundamental laws of nature. This production function also includes possible feedback effects of wastes on production, arising through the ambient levels of pollutants. It is, however, relatively uncommon for such a fully specified production function to be used in either theoretical or empirical work in economics. In particular cases, this could be justified by argument that for the purpose at hand – examining the implications of resource depletion, say – nothing essential is lost by an incomplete specification, and the analysis is simplified and clarified. We shall implicitly use this argument ourselves at various points in the remainder of the book, and work with specialised versions of equation 2.5. However, it is important to keep in mind that it is equation 2.5 itself that is the correct specification of a production process that has a material output.

2.1.3.3 Ecology

Ecology is the study of the distribution and abundance of plants and animals. A fundamental concept in ecology is the ecosystem, which is an interacting set of plant and animal populations, together with their abiotic, i.e. non-living, environment. An ecosystem can be defined at various scales from the small and local – a pond or field – through to the large and global – the biosphere as a whole.

2.1.3.3.1 Stability and resilience

Two concepts of fundamental importance in ecology are stability and resilience. The ecologist Holling (1973, 1986) distinguishes between stability as a property attaching to the populations comprising an ecosystem, and resilience as a property of the ecosystem. Stability is the propensity of a population to return to some kind of equilibrium following a disturbance. Resilience is the propensity of an ecosystem to retain its functional and organisational structure following a disturbance. The fact that an ecosystem is resilient does not necessarily imply that all of its component populations are stable. It is possible for a disturbance to result in a population disappearing from an ecosystem, while the ecosystem as a whole continues to function in broadly the same way, so exhibiting resilience.

Common and Perrings (1992) put these matters in a slightly different way. Stability is a property that relates to the levels of the variables in the system. Cod populations in North Atlantic waters would be stable, for example, if their numbers returned to prior levels after a brief period of heavy fishing was brought to an end. Resilience relates to the sizes of the parameters of the relationships determining ecosystem structure and function in terms, say, of energy flows through the system. An ecosystem is resilient if those parameters tend to remain unchanged following shocks to the system, which will mean that it maintains its organisation in the face of shocks to it, without undergoing catastrophic, discontinuous, change.

Some economic activities appear to reduce resilience, so that the level of disturbance to which the ecosystem can be subjected without parametric change taking place is reduced. Expressed another way, the threshold levels of some system variable, beyond which major changes in a wider system take place, can be reduced as a consequence of economic behaviour. Safety margins become tightened, and the integrity and stability of the ecosystem is put into greater jeopardy. This aligns with the understanding, noted above, of pollution as that which occurs when a waste flow exceeds the assimilative capacity of the receiving system, and that which if it occurs itself reduces the system's assimilative capacity.

When such changes takes place, dose–response relationships may exhibit very significant non-linearities and discontinuities. Another way of putting this is to say that dose–response relationships may involve thresholds. Pollution of a water system, for example, may have relatively small and proportional effects at low pollution levels, but at higher pollutant levels, responses may increase sharply and possibly jump discontinuously to much greater magnitudes. Such a dose–response relationship is illustrated in Figure 2.3.

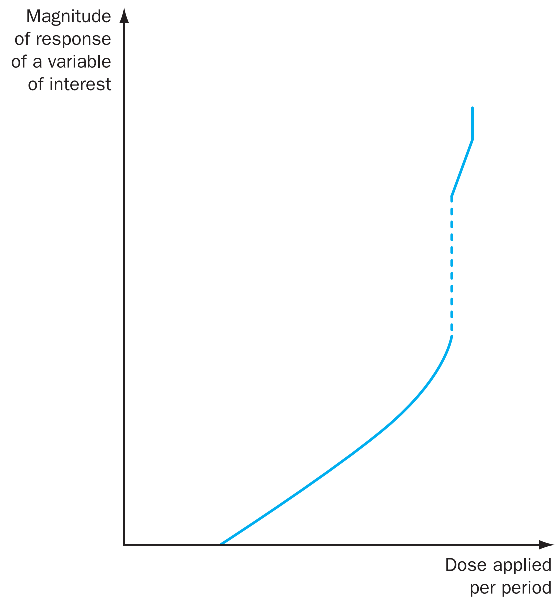


Figure 2.3 Non-linearities and discontinuities in dose–response relationships

2.1.3.3.2 The ecological impact of humanity

From an ecological perspective, humanity is one animal species among many. Animals feed on plants, and, in some cases, other animals. Box 2.1 reports estimates of humanity's global impact in terms of that species' appropriation of the plant food available to all animals.

Box 2.1 Human appropriation of the products of photosynthesis

The basis for life on earth is the capture by plants of radiant solar energy, and its conversion to organic material by the process of photosynthesis. The rate at which plants produce plant tissue is primary productivity, measured in terms of energy per unit area per unit time – calories per square metre per year, say. Gross primary productivity is the total amount of solar energy that is fixed by photosynthesis, whereas net primary productivity is that less the amount of energy lost to the environment as respiration, and so the amount that is actually stored in the plant tissue. Net primary productivity is the measure of the energy that is potentially available to the animals that eat the plants.

Table 2.1 shows estimates of the proportion of net primary productivity that is appropriated by humanity. About 70% of the earth's surface is covered by water. The aquatic zone produces about 40% of total global net primary productivity. The terrestrial zone, although accounting for only 30% of the surface area, accounts for about 60% of total primary productivity.

For each zone, and for both zones together, Table 2.1 shows estimates of human appropriation on three different bases:

- Low – for this estimate what is counted is what humans and their domesticated animals directly use as food, fuel and fibre.
- Intermediate – this counts the current net primary productivity of land modified by humans. Thus, for example, whereas the low estimate relates to food eaten, the intermediate estimate is of the net primary productivity of the agricultural land on which the food is produced.
- High – this also counts potential net primary productivity that is lost as a result of human activity. Thus, with regard to agriculture, this estimate includes what is lost as a result, for example, of transforming forested land into grassland pasture for domesticated animals. It also includes losses due to desertification and urbanisation.

Table 2.1 Human appropriation of net primary productivity

	Percentages		
	Low	Intermediate	High
Terrestrial	4	31	39
Aquatic	2	2	2
Total	3	19	25

Source: Vitousek *et al.* (1986)

For the aquatic zone, it makes no difference which basis for estimation is used. This reflects the fact that human exploitation of the oceans is much less than it is of land-based ecosystems, and that the former is still essentially in the nature of hunter–gatherer activity rather than agricultural activity. It also reflects that what are reported are rounded numbers, to reflect the fact that we are looking at – for both zones – approximations rather than precise estimates.

For the terrestrial zone, the basis on which the human appropriation of net primary productivity is measured makes a lot of difference. If we look at what humans and their domesticates actually consume – the low basis – it is 4%. If we look at the net primary productivity of land managed in human interests – the intermediate basis – it is 31%. Commenting on the high terrestrial figure, the scientists responsible for these estimates remark:

> An equivalent concentration of resources into one species and its satellites has probably not occurred since land plants first diversified.
> (Vitousek *et al.*, 1986, p. 372)

Rojstaczer *et al.* (2001) report the results of a similar study using a wider range of more recent data, confining their attention to terrestrial net primary production and looking only at the intermediate basis for estimation. As well as reporting the mean estimate for human appropriation, Rojstaczer *et al.* used Monte Carlo methods to assess the range of uncertainty implied by the available data. Their mean estimate of the proportion of terrestrial net primary production appropriated by humans is 32%, which is almost exactly the same as that of Vitousek *et al.*, 31% in Table 2.1 here, for intermediate terrestrial appropriation. The 95% confidence limits reported by Rojstaczer *et al.* are 32% ± 22%, i.e. a lower bound of 10% and an upper bound of 54%. They comment:

> Although there is a large degree of uncertainty, it is clear that human impact on TNPP [terrestrial net primary production] is significant. The lower bound on our estimate . . . indicates that humans have had more impact on biological resources than any single species of megafauna known over the history of the earth.

2.1.3.3.3 Ecological footprints

The quotation which introduces this chapter refers to 'humanity's ecological footprint'. This is another way of expressing the ecological impact of the human species. An ideal definition (Wackernagel and Rees, 1997) of a particular human economy's ecological footprint is:

> the aggregate area of land and water in various ecological categories that is claimed by participants in the economy to produce all the resources they consume, and to absorb all the wastes they generate on a continuing basis, using prevailing technology.

We describe this as an 'ideal' definition because to date estimates of the size of ecological footprints have been based on just subsets of consumed resources and generated wastes, and are in that sense conservative estimates. It should also be noted that the footprint size will vary with technology as well as with levels and patterns of production and consumption. Wackernagel *et al*. (2002) report estimates of the size of the footprint for each of the years from 1961 to 1999, for the whole global economy. Considering the demands for land and water on account of

- growing crops
- grazing domesticated animals
- harvesting timber
- fishing
- space for locating human artefacts such as houses, factories, roads, etc.
- sequestering the CO_2 released in fossil-fuel combustion

in relation to the available amounts in the biosphere, they find that for all of humanity the ratio of the former demand to the latter supply increased from approximately 0.7 in 1961 to approximately 1.2 in 1999, and conclude that as presently constituted the global economy is not sustainable in that it would 'require 1.2 earths, or one earth for 1.2 years, to regenerate what humanity used in 1999'. For 2003 the global human ecological footprint was 1.25 (from http://www.footprintnetwork.org/, May 2008). On a per capita basis the global average demand for biologically productive space in 2003 was 2.3 hectares, whereas other studies have estimated per capita footprints of 9.7 hectares for the USA, 5.4 for the UK and 4.7 for Germany. The implication is that if the developing world were to attain the consumption levels of the developed world, using current technology, the total footprint for the world would be the size of several earths.

2.1.3.3.4 Biodiversity

A definition of biodiversity is:

> the number, variety and variability of all living organisms in terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are parts.[2]

It is evident from this definition that biodiversity is intended to capture two dimensions: first, the number of biological organisms and, second, their variability. There are three levels at which biodiversity can be considered:

1. Population: genetic diversity within the populations that constitute a species is important as it affects evolutionary and adaptive potential of the species, and so we might measure biodiversity in terms of the number of populations.
2. Species: we might wish to measure biodiversity in terms of the numbers of distinct species in particular locations, the extent to which a species is endemic (unique to a specific location), or in terms of the diversity (rather than the number) of species.
3. Ecosystems: in many ways, the diversity of ecosystems is the most important measure of biodiversity; unfortunately, there is no universally agreed criterion for either defining or measuring biodiversity at this level.

For the purposes of this classification of levels, a species can be taken to be a set of individual organisms which have the capacity to reproduce, while a population is a set that actually do reproduce. A population is, that is, a reproductively isolated subset of a species.

[2] This definition is taken from the Convention on Biological Diversity adopted at the UNCED conference in Rio de Janeiro in 1992: see 2.5.2 below.

Biodiversity is usually considered in terms of species, and the number of distinct species is often used as the indicator of biodiversity. There are problems with this measure. For example, within one population of any species there will be considerable genetic variation. Suppose a harvesting programme targets individuals within that population with a particular characteristic (such as large size). The target individuals are likely to possess genetic material favouring that characteristic, and so the harvesting programme reduces the diversity of the gene pool in the remaining population. Managed harvesting programmes, therefore, may result in loss of biodiversity even though the number of extant species shows no change.

Biodiversity is important in the provision of environmental services to economic activity in a number of ways. In regard to life-support services, diverse ecological systems facilitate environmental functions, such as carbon cycling, soil fertility maintenance, climate and surface temperature regulation, and watershed flows. The diversity of flora and fauna in ecosystems contributes to the amenity services that we derive from the environment. In relation to inputs to production, those flora and fauna are the source of many useful products, particularly pharmaceuticals, foods and fibres; the genes that they contain also constitute the materials on which future developments in biotechnology will depend. In terms of agriculture, biodiversity is the basis for crop and livestock variability and the development of new varieties.

Ecologists see the greatest long-term importance of biodiversity in terms of ecosystem resilience and evolutionary potential. Diverse gene pools represent a form of insurance against ecological collapse: the greater is the extent of diversity, the greater is the capacity for adaptation to stresses and the maintenance of the ecosystem's organisational and functional structure.

We have very poor information about the current extent of biodiversity. The number of species that currently exist is not known even to within an order of magnitude. Estimates that can be found in the literature range from 3–10 million (May, 1988) to 50–100 million (Wilson, 1992). A current best guess of the actual number of species is 12.5 million (Groombridge, 1992). Even the currently known number of species is subject to some dispute, with a representative figure being 1.7 million species described to date (Groombridge, 1992). About 13 000 new species are described each year. Table 2.2 reports current knowledge about species numbers for a variety of important taxonomic classes.

Table 2.2 Numbers of described species and estimates of actual numbers for selected taxa (thousands)

Taxa	Species described	Estimated number of species: high	Estimated number of species: low	Working figure
Viruses	4	1 000	50	400
Bacteria	4	3 000	50	1000
Fungi	72	2 700	200	1500
Protozoa and algae	80	1 200	210	600
Plants	270	500	300	320
Nematodes (worms)	25	1 000	100	400
Insects	950	100 000	2000	8000
Molluscs	70	200	100	200
Chordates	45[a]	55	50	50

Source: Jeffries (1997, p. 88), based in turn on Groombridge (1992) and Heywood (1995)

[a] Of the 45 000 chordates (vertebrate animals), there are about 4500 mammals, 9700 birds, 4000 amphibians and 6550 reptiles

2.1.3.3.5 The Millennium Ecosystem Assessment

The Millennium Ecosystem Assessment, MEA, was called for by the UN Secretary-General in 2000. It was conducted over 2001 to 2005, being coordinated by the United Nations Environment Programme. The MEA was intended to assess the implications for human well-being of ecosystem change, and to establish the scientific basis for actions to enhance the conservation and sustainable use of ecosystems and their contribution to human well-being. It proceeded on the basis of synthesising existing information, rather than seeking to generate new data. It involved some 2000 scientists. The results of this huge undertaking are reported in several volumes, which are available as books and for downloading from the MEA website, the address for which is http://www.millenniumassessment.org/en/index.aspx.

The four main findings of the MEA are reported in the Summary for Decision-Makers section of the overall synthesis volume (Millennium Ecosystem Assessment, 2005a) as follows:

Box 2.2 Biodiversity loss and human impact

For ecologists, the appropriation of the products of photosynthesis described in Box 2.1 is the most fundamental human impact on the natural environment, and is the major driver of the current high rate of biodiversity loss. In a speech at the Natural History Museum on 28 November 2001, the ecologist Lord Robert May, President of the Royal Society and formerly the UK government's chief scientist, stated that:

> There is little doubt that we are standing on the breaking tip of the sixth great wave of extinction in the history of life on earth. It is different from the others in that it is caused not by external events, but by us – by the fact that we consume somewhere between a quarter and a half of all the plants grown last year.
>
> (Quoted in *The Guardian*, 29 November 2001)

Just how fast is the stock of genetic resources being depleted? Given that the number of species existing is not known, statements about rates of extinction are necessarily imprecise, and there are disagreements about estimates. Table 2.3 shows data for *known* extinctions since 1600. The actual number of extinctions would certainly be equal to or exceed this. The recorded number of extinctions of mammal species since 1900 is 20. It is estimated from the fossil record that the normal, long-run average, rate of extinction for mammals is one every two centuries. In that case, for mammals the known current rate of extinction is 40 times the background rate.

Table 2.3 Species extinctions since 1600

Kingdom	Extinct species
Vertebrates	**337**
Mammals	87
Birds	131
Reptiles	22
Amphibians	5
Fishes	92
Invertebrates	**389**
Insects	73
Molluscs	303
Crustaceans	9
Others	4
Plants	**90**
Mosses	3
Conifers, Cycads, etc.	1
Flowering plants	86

Source: Table 1.4 Convention on Biological Diversity 2001 (access at http://www.cbd.int/gbo1/chap-01-02.shtml)

To quote Lord Robert May again:

> If mammals and birds are typical, then the documented extinction rate over the past century has been running 100 to more like 1000 times above the average background rate in the fossil record. And if we look into the coming century it's going to increase. An extinction rate 1000 times above the background rate puts us in the ballpark of the acceleration of extinction rates that characterised the five big mass extinctions in the fossil records, such as the thing that killed the dinosaurs.
>
> (*The Guardian*, 29 November 2001)

According to Wilson (1992) there could be a loss of half of all extant birds and mammals within 200–500 years. For all biological species, various predictions suggest an overall loss of between 1% and 10% of all species over the next 25 years, and between 2% and 25% of tropical forest species (UNEP, 1995). In the longer term it is thought that 50% of all species will be lost over the next 70 to 700 years (Smith *et al.*, 1995; May, 1988).

Lomborg (2001) takes issue with most of the estimates of current rates of species loss made by biologists. His preferred estimate for the loss of animal species is 0.7% per 50 years, which is smaller than many of those produced by biologists. It is, however, in Lomborg's own words: 'a rate about 1500 times higher than the natural background extinction' (p. 255). There really is no disagreement about the proposition that we are experiencing a wave of mass extinctions, and that it is due to the human impact on the environment.

Over the past 50 years, humans have changed ecosystems more rapidly and more extensively than in any comparable period of human history, largely to meet rapidly growing demands for food, fresh water, timber, fiber, and fuel. This has resulted in a substantial and largely irreversible loss in the diversity of life on earth.

The changes that have been made to ecosystems have contributed to substantial net gains in human well-being and economic development, but these gains have been achieved at growing cost in the form of the degradation of many ecosystem services, increased risk of nonlinear changes, and the exacerbation of poverty for some groups of people. These problems, unless addressed, will substantially diminish the benefits that future generations obtain from ecosystems.

The degradation of ecosystem services could grow significantly worse during the first half of this century and is a barrier to achieving the

> Millennium Development Goals. (See section 2.5.4 at the end of this chapter for discussion of these goals.)
>
> The challenge of reversing the degradation of ecosystems while meeting increasing demands for their services can be partially met under some scenarios that the MA has considered, but these involve significant changes in policies, institutions and practices that are not currently under way. Many options exist to conserve or enhance specific ecosystem services in ways that reduce negative trade-offs or that provide positive synergies with other ecosystem services.

In regard to biodiversity loss, the MEA estimates that the rate of known extinctions in the past century was 50–500 times greater than the 'normal' extinction rate calculated from the fossil record, which is 0.1–1 extinctions per 1000 species per 1000 years. If species that have possibly gone extinct in the past 100 years are included, the extinction rate for the past century is 'up to 1000 times higher than the background extinction rates' as calculated from the fossil record. For more information on the MEA of the situation in regard to biodiversity loss see the biodiversity synthesis volume (Millennium Ecosystem Assessment, 2005b).

The major cause of the acceleration in the extinction rate is, according to the MEA, the appropriation of the products of photosynthesis by the human species. Figure 3 in the overall synthesis volume (Millennium Ecosystem Assessment, 2005a) reports estimates of the proportions of 13 (out of a total of 14) of the world's terrestrial biomes that our species has converted to use for the human species. A biome is the largest unit of ecological classification, and comprises many inter-connected ecosystems. For 4 biomes – mediterranean forests, woodlands and scrub; temperate forest steppe and woodland; temperate broadleaf and mixed forests; tropical and sub-tropical dry broadleaf forests – the percentage already converted exceeds 50% (and for the first two is around 70%). It is estimated that for three more – flooded grasslands and savannas; tropical and sub-tropical grasslands, savannas and shrublands; tropical and sub-tropical coniferous forests – the proportion converted will exceed 50%, and approach 70%, by 2050. These seven biomes are the most productive, in terms of photosynthetic conversion.

2.2 The drivers of environmental impact

The environmental impact of economic activity can be looked at in terms of extractions from or insertions into the environment. In either case, for any particular instance the immediate determinants of the total level of impact are the size of the human population and the per capita impact. The per capita impact depends on how much each individual consumes, and on the technology of production. This is a very simple but useful way to start thinking about what drives the sizes of the economy's impacts on the environment. It can be formalised as the IPAT identity.

2.2.1 The IPAT identity

The IPAT identity is

$$I \equiv P \times A \times T \tag{2.6}$$

where

I is impact, measured as mass or volume
P is population size
A is per capita affluence, measured in currency units
T is technology, as the amount of the resource used or waste generated per unit production

Let us look at impact in terms of mass, and use GDP for national income. Then T is resource or waste per unit GDP. Then for the resource extraction case, the right-hand side of (2.6) is

$$\text{Population} \times \frac{\text{GDP}}{\text{Population}} \times \frac{\text{Resource use}}{\text{GDP}}$$

where cancelling the two population terms and the two GDP terms leaves Resource use, so that (2.6) is an identity. If mass in measured in tonnes, GDP in \$, and population is n, we have

$$\text{tonnes} \equiv n \times \frac{\$}{n} \times \frac{\text{tonnes}}{\$}$$

where on the right-hand side the ns and the \$s cancel to leave tonnes.

The IPAT identity decomposes total impact into three multiplicative components – population,

Table 2.4 Global carbon dioxide scenarios

	P (billions)	A (PPP US$)	T (tonnes per $)	I (billions of tonnes)
Current	6.5148	9 543	0.0004662	28.9827
$P \times 1.5$	9.7722	9 543	0.0004662	32.3226
$P \times 1.5$ and $A \times 2$	9.7722	19 086	0.0004662	86.9520
$P \times 1.5$ and $A \times 2$ with I at current	9.7722	19 086	0.0001554	28.9827

Source: UNDP (2007) Tables 1, 5 and 24

affluence and technology. To illustrate the way in which IPAT can be used, consider global carbon dioxide emissions. The first row of Table 2.4 shows the current situation. The first-row figures for P, A as 2005 world GDP per capita in 2005 PPP US$, and I as 2004 global carbon dioxide emissions are taken from the indicated source: the figure for T is calculated by dividing I by P times A to give tonnes of carbon dioxide per $ of GDP.[3] The second row uses the T figure from the first to show the implications for I of a 50% increase in world population, for constant affluence and technology. The third row also uses the T figure from the first to show the implications of that increase in population together with a doubling of per capita GDP. A 50% increase in world population is considered because that is a conservative round number for the likely increase to 2100 (see below), and a doubling of per capita GDP is used as a round-number conservative estimate of what would be necessary to eliminate poverty (see below). As will be discussed in Chapter 9, many climate experts take the view that the current level of carbon dioxide emissions is dangerously high. The fourth row in Table 2.4 solves IPAT for T when I is set equal to its level in the first row, and P and A are as in the third row – compared with the first-row figure for T, it shows that carbon dioxide emissions per unit GDP would have to be reduced to one-third of their current level in order to keep total emissions at their current level given a 50% population increase and a doubling of affluence.

We now look briefly at the current situation, a little history, and future prospects in regard to each of P, A and T.

2.2.2 Population

At the time that this chapter was being written, April 2008, the most recent year for which data on global population size was available was 2005. In that year the estimated human population was 6.5148 billion. The estimated growth rate for 1975–2005 was 1.6% per year. The staggering increase in human population in the second half of the twentieth century can be gauged by the fact that in 1950 world population was 2.5 billion – it more than doubled over 50 years to 6 billion in 2000. At the beginning of the nineteenth century the world's population is estimated to have been about 0.9 billion.

The projections for the global human population shown in Figure 2.4 are taken from the Executive Summary of a publication of the Population Division of the UN's Department of Economic and Social Affairs (UN Population Division, 2000). They differ according to the assumptions made about fertility. The medium projection assumes that fertility in all major areas of the world stabilises at the replacement level around 2050. The low projection assumes that fertility is half a child lower than for medium, and the high projection half a child higher. As Figure 2.4 shows, the long-run prospects for the size of the human population are very sensitive to what is assumed about future fertility – the vertical axis units are millions. All the projections make the same assumption about longevity.

[3] PPP stands for purchasing power parity. In making international GDP comparisons, and aggregating GDP across countries, using market exchange rates overlooks the fact that average price levels differ across countries, and are generally lower in poor countries. Using market exchange rates exaggerates differences in real income levels. Purchasing power parity exchange rates, relative to the US$, are calculated by the International Comparison Programme in order to overcome this problem, and PPP US$ GDP data convert local-currency GDP at these exchange rates: see UN Statistical Division (1992). The difference between the market exchange-rate GDP figure and the PPP exchange-rate figure can be large – for China in 1999, for example, the latter was more than four times the former. Whereas on the former basis the US economy was nine times as big as the Chinese economy, on the latter basis it was twice as big. In terms of carbon dioxide emissions per unit GDP, on the market exchange-rate measure of GDP the Chinese figure for T is more than five times that for the US; on the PPP exchange-rate basis of GDP measurement it was just 25% bigger.

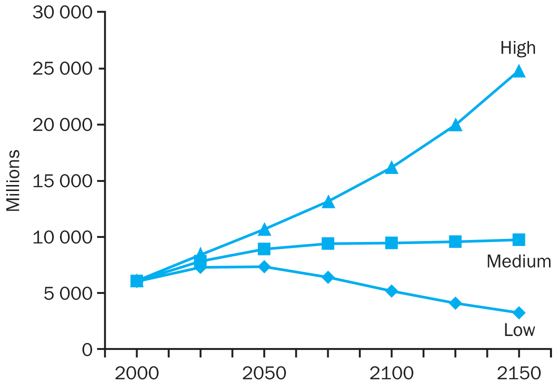


Figure 2.4 World population projections
Source: Data from UN Population Division (2000)

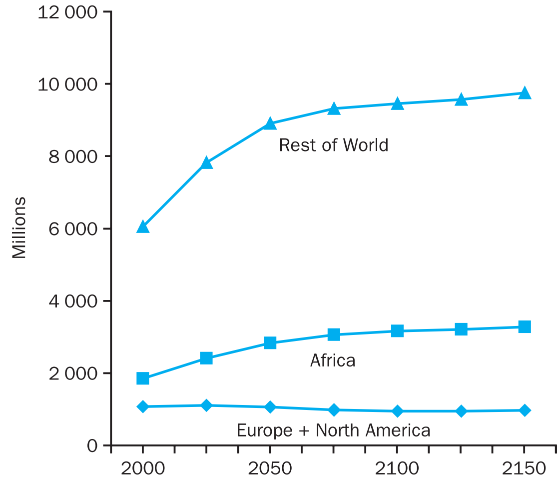


Figure 2.5 Contributions to world population growth to 2150
Source: Data from UN Population Division (2000)

The percentage rate of increase of global population is already well below its historical peak, having decreased in recent years in all regions of the world. Growth rates currently average less than 1% per year in developed countries (0.8% over 1975–2005 for the OECD) and less than 2% in developing countries (1.9% over 1975–2005 for developing as defined by the UNDP).[4] In many countries (including most OECD countries and China), fertility rates are below the replacement rates that are required for a population size to be stationary in the long run. For these countries, population is destined to fall at some point in the future even though the momentum of population dynamics implies that population will continue to rise for some time to come. For example, although the Chinese birth-rate fell below the replacement rate in 1992, population is projected to rise from 1.3 billion in 2005 to 1.5 billion by 2050, on the medium UN scenario discussed above.

Differences in fertility, and longevity, rates as between different parts of the world mean that the distribution of the world population as between different regions will change. Figure 2.5, based on data from the same source as used for Figure 2.4, illustrates. It relates to the medium scenario. The lower line shows the combined population of Europe and North America more or less constant in absolute terms, and so falling as a proportion of the total (from about 18% now to about 10% in 2150). The gap between the lower and middle lines shows what is happening to the population of Africa – it grows absolutely and as a proportion of the world total (from about 13% now to about 24% in 2150).

2.2.3 Affluence

As shown in Table 2.4, the 1999 world average for GDP per capita, in round numbers of 2005 PPP US$, was 9500. To get some sense of what this means, note the following figures (also from UNDP, 2007) for 2005 GDP per capita in 2005 PPP US$ for a few selected individual nations:

USA	41 890
UK	33 238
Germany	29 461
Czech Republic	20 538
Portugal	20 410
Hungary	17 887
China	6 757
India	3 452
Kenya	1 240
Sierra Leone	806

The world average is more than twice that for India, and about 20% of that for the USA.

[4] The figures here are taken from UNDP (2007), Table 5.

Over the period 1975 to 2005, world average GDP per capita grew at 1.4% per annum. At that rate of growth, over 50 years the level of world average GDP would just about double, taking it to about the current level for the Czech Republic. A longer-term perspective is provided in Maddison (1995), where per capita GDP, in 1990 PPP $s, is estimated for 57 countries from 1820 through to 1992. For this sample of countries, which currently account for over 90% of world population, mean estimated per capita GDP grew from about $1000 in 1820 to about $8500 in 1992. Notwithstanding the necessary imprecision in estimates of this kind, it is clear that over the past two centuries, average global affluence has increased hugely. It is also clear that it is currently distributed very unevenly – a matter to which we return below.

2.2.4 Technology

Given the range of things that we extract from and insert into the environment, even a summary documentation of values for *T* as mass extracted, or inserted, per $ of economic activity would be very long, and well beyond the scope of this book. One way of giving some summary sense of the role of technology in environmental impact is to look at energy use. There are three reasons for this. First, energy is the potential to do work and energy use increases with work done. Moving and transforming matter requires work, and the amount of energy used directly reflects the amount of movement and transformation. It is the levels of extractions and insertions by the economy that determine its environmental impact, and those levels, which are linked by the law of conservation of mass, are measured by the level of energy use. While it is true that some extractions and insertions are more damaging than others, the level of its energy use is a good first approximation to the level of an economy's environmental impact.

The second and third reasons both follow from the fact that in the modern industrial economies that now dominate the global economy, about 90% of energy use is based on the combustion of the fossil fuels – coal, oil, gas. These are non-renewable resources where recycling is impossible. Hence the second reason for looking at energy – the more we use now, the less fossil fuel resources are available to future generations. The third reason is that fossil fuel combustion is directly a major source of insertions into the environment, and especially the atmosphere. Particularly, about 80% of carbon dioxide emissions originate in fossil fuel combustion, and carbon dioxide is the most important of the greenhouse gases involved in the enhanced greenhouse effect.

The energy that an animal acquires in its food, and which is converted into work, growth and heat, is called somatic energy. When the human animal learned how to control fire, about 500 000 years ago, it began the exploitation of extrasomatic energy. It began, that is, to be able to exert more power than was available from its own muscles. The human energy equivalent, HEE, is a unit of measure which is the amount of somatic energy required by a human individual. This amount varies across individuals and with circumstances. A convenient amount to use for the HEE is 10 megajoules per day, which is a round-number version of what is required by an adult leading a moderately active life in favourable climatic conditions.

Human history can be divided into three main phases, the distinguishing characteristics of which are technological. The first two phases are distinguished according to the technology for food production. The first is the hunter–gatherer phase, which lasted from the beginning of human history until about 12 000 years ago – it accounts for most of human history. During this phase food production involved gathering wild plants and hunting wild animals. It is estimated that the use of fire by an individual in hunter–gatherer societies was, on average and approximately, equivalent to the use of 1 HEE – per capita the use of fire was about equivalent to the amount of energy flowing through a human body. The total per capita use of energy was, that is, about 2 HEE.[5]

The agricultural phase of human history lasted about 12 000 years, and ended about 200 years ago.

[5] The estimates for HEE for hunter–gatherers and agriculturalists here are taken from Boyden (1987), as is the global average figure for 1900 given below. The HEE estimates for 1997 are based on data from WRI (2000).

Agriculture involves producing food by domesticating some plant and animal species, and managing the environment so as to favour those species as against wild species. The technology of energy use was evolving throughout the agricultural phase of history. By its end the average human being was deploying some 3–4 HEE, so that in addition to her own muscle power she was using extrasomatic energy at the rate of 2–3 HEE. In addition to fire, almost entirely based on biomass (mainly wood) combustion, the sources of extrasomatic energy were animal muscles, the wind, and water. Animals – horses, oxen, donkeys – were used mainly for motive power in transport and agriculture. The wind was used to propel boats, to drive pumps for lifting water, and to drive mills for grinding corn. Water mills were also used for grinding corn, as well as powering early machinery for producing textiles and the like.

Comparing the situation at the end of the agricultural phase of human history with that of the hunter–gatherer phase, the per capita use of energy had approximately doubled, and the population size had increased by a factor of about 200, so that total energy use by humans had increased by a factor of about 400.

The industrial phase of human history began about 200 years ago, around 1800. Its distinguishing characteristic has been the systematic and pervasive use of the fossil fuels. In the first instance this was mainly about the use of coal in manufacturing, and then in transport. In the twentieth century oil use became much more important, as did the use of it, as well as coal, to produce electricity. In the twentieth century, the use of fossil fuels and electricity became standard, in the more advanced economies, in the domestic household sector, and in agricultural production. In a modern economy, nothing is produced that does not involve the use of extrasomatic energy, and most of what is used is based on fossil fuel combustion.

By 1900 the average human used about 14 extrasomatic HEE. By the end of the twentieth century the average human used about 19 extrasomatic HEE – the equivalent of 19 human slaves. This global average for 1997 comes from a wide range for individual nations. In 1997, per capita extrasomatic energy use in the USA was 93 HEE, while in Bangladesh it was 4 (mainly from biomass). Comparing the situation at the end of the twentieth century with that at the end of the eighteenth, the human population had increased in size by a factor of approximately 6, while extrasomatic energy use per capita had also increased by a factor of approximately 6. In 200 years total global extrasomatic energy use had increased by a factor of about 35. As noted above, this implies that the work done in moving and transforming matter – the scale of economic activity and its impact on the environment – had increased by a factor of 35.

2.2.5 Behavioural relationships

IPAT is an accounting identity. Given the way that *P*, *A* and *T* are defined and measured, it must always be the case that *I* is equal to *PAT*. As we saw, IPAT can be useful for figuring the implications of certain assumptions, for producing scenarios. In Table 2.4 we used it, for example, to calculate *I* on the assumption that *P* increased by 50%, *A* increased by 100%, and *T* remained the same. *P*, *A* and *T* are the proximate drivers of *I*. But we could ask, what drives *P*, *A* and *T*? Apart from being an interesting question, this is important if we want to consider policies to drive some *I*, such as carbon dioxide emissions for example, in a particular direction. We could, that is, look to build a model which incorporates the behavioural relationships that we think determine what happens to *P*, *A* and *T*, and other variables, over time. In such a model we would very likely have relationships between *P*, *A* and *T*, as well as between them and other variables.

There are many behavioural relationships that affect, and are affected by, movements in *P*, *A* and *T*. Economists are particularly interested, for example, in supply and demand functions for inputs to production. These determine the relative prices of those inputs, and hence affect *T* – a high price for fossil fuels will reduce their use, and hence reduce carbon dioxide emissions. Much of the rest of the book will be concerned with the role of the price mechanism in relation to the determination of the level of extractions from and insertions into the natural environment. Here we will look at two examples of behavioural relationships where affluence is the driver.

2.2.5.1 Affluence and population growth: the demographic transition

A statistical relationship that is often remarked upon is the negative correlation between income level and population growth rate. Several attempts have been made to explain this observed relationship, the most well known of which is the theory of demographic transition (Todaro, 1989). The theory postulates four stages through which population dynamics progress, shown in Figure 2.6. In the first stage, populations are characterised by high birth-rates and high death-rates. In some cases, the death-rates reflect intentions to keep populations stable, and so include infanticide, infant neglect and senilicide (see Harris and Ross, 1987). In the second stage, rising real incomes result in improved nutrition and developments in public health which lead to declines in death-rates and rapidly rising population levels. In the third stage of the demographic transition, economic forces lead to reduced fertility rates. These forces include increasing costs of childbearing and family care, reduced benefits of large family size, higher opportunity costs of employment in the home, and changes in the economic roles and status of women. In the final stage, economies with relatively high income per person will be characterised by low, and approximately equal, birth- and death-rates, and so stable population sizes.

The theory of demographic transition succeeds in describing the observed population dynamics of many developed countries quite well. If the theory were of general applicability, it would lead to the

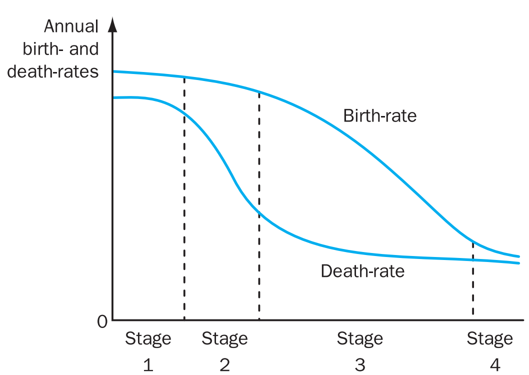
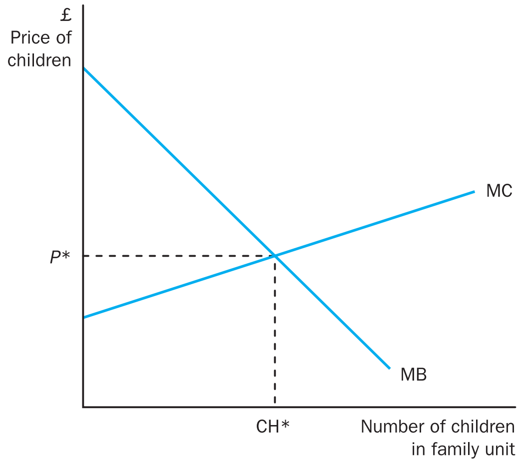


Figure 2.6 The theory of demographic transition

conclusions that rising population is a transient episode, and that programmes which increase rates of income growth in developing countries would lower the time profile of world population levels. But it remains unclear whether the theory does have general applicability. For many developing countries the second stage was reached not as a consequence of rising real income but rather as a consequence of knowledge and technological transfer. In particular, public health measures and disease control techniques were introduced from overseas at a very rapid rate. The adoption of such measures was compressed into a far shorter period of time than had occurred in the early industrialising countries, and mortality rates fell at unprecedented speed. During the nineteenth century, the higher-income countries typically experienced falls in birth-rates relatively soon after falls in mortality rates. However, while birth-rates are falling in most developing countries, these falls are lagging behind drops in the mortality levels, challenging the relevance of the theory of demographic transition. Dasgupta (1992) argues that the accompanying population explosions created the potential for a vicious cycle of poverty, in which the resources required for economic development (and so for a movement to the third stage of the demographic transition) were crowded out by rapid population expansion.

Two important determinants of the rate at which a population changes over time are the number of children born to each female of reproductive age, and the life expectancy of each child. There have been dramatic increases in life expectancy throughout the world, attributed to improved medical and public health services. The number of children born into each household is primarily the outcome of a choice made by (potential) parents. Family size is the choice-variable; contraceptive and other family planning practices are the means by which that choice is effected. Microeconomic theory suggests that the marginal costs and the marginal benefits of children within the family (see Figure 2.7) determine family size. The marginal costs of children depend on the costs of childbearing, child rearing and education, including the opportunity costs of parental time in these activities. Marginal benefits of children to the family include the psychic benefits of children, the contribution of children to family income, and



MC = The marginal cost to the family of a child
MB = The marginal benefit to the family of a child
= (the demand curve for children)

Figure 2.7 The microeconomics of fertility

the extent to which security in old age is enhanced by larger family size.

An important advantage of this line of analysis is that it offers the prospect of deriving guidelines for population policy: attempts to alter desired family size should operate by shifting the marginal cost of bearing and raising children, or the marginal benefits derived from children within the family. What measures might governments take, or what intermediate goals might they pursue, to reduce the desired family size? Several suggest themselves:

- Increased levels of education, particularly education of women. This could affect fertility through three related routes. First, education enhances the effectiveness of family planning programmes: families become more proficient at having the number of children that they choose. Secondly, greater participation in education increases the status of women: it is now widely agreed that where females have low-status roles in the culture of a society, fertility rates are likely to be high. Thirdly, greater education decreases labour-market sex discrimination, allows females to earn market incomes, and raises real wage rates in the labour market. These changes increase the opportunity cost of children, and may well also reduce the marginal benefits of children (for example, by salaried workers being able to provide for old age through pension schemes).
- Financial incentives can be used to influence desired family size. Financial penalties may be imposed upon families with large numbers of children. Alternatively, where the existing fiscal and welfare state provisions create financial compensation for families with children, those compensations could be reduced or restructured. There are many avenues through which such incentives can operate, including systems of tax allowances and child benefits, subsidised food, and the costs of access to health and educational facilities. There may well be serious conflicts with equity if financial incentives to small family size are pushed very far, but the experiences of China suggest that if government is determined, and can obtain sufficient political support, financial arrangements that increase the marginal cost of children or reduce the marginal benefits of children can be very powerful instruments.
- Provision of care for and financial support of the elderly, financed by taxation on younger groups in the population. If the perceived marginal benefits of children to parents in old age were to be reduced (by being substituted for in this case), the desired number of children per family would fall. As the tax instrument merely redistributes income, its effect on welfare can be neutral. But by reducing the private marginal benefits of children it can succeed (at little or no social cost) in reducing desired family size.
- The most powerful means of reducing desired family size is almost certainly economic development, including the replacement of subsistence agriculture by modern farming practices, giving farm workers the chance of earning labour market incomes. There may, of course, be significant cultural losses involved in such transition processes, and these should be weighed against any benefits that agricultural and economic development brings. Nevertheless, to the extent that subsistence and non-market farming dominates an economy's agricultural sector, there will be powerful incentives for large family size. Additional children are valuable assets to the family, ensuring that the perceived

marginal benefits of children are relatively high. Furthermore, market incomes are not being lost, so the marginal cost of child-rearing labour time is low. Important steps in the direction of creating markets for labour (and reducing desired family size) can be taken by defining property rights more clearly, giving communities greater control over the use of local resources, and creating financial incentives to manage and market resources in a sustainable way.

2.2.5.2 Affluence and technology: the EKC

The *World Development Report 1992* (World Bank, 1992) was subtitled 'Development and the environment'. It noted that 'The view that greater economic activity inevitably hurts the environment is based on static assumptions about technology, tastes and environmental investments'. If we consider, for example, the per capita emissions, e, of some pollutant into the environment, and per capita income, y, then the view that is being referred to can be represented as

$$e = \alpha y \tag{2.7}$$

so that e increases linearly with y, as shown in Figure 2.8(a). Suppose, alternatively, that the coefficient α is itself a linear function of y:

$$\alpha = \beta_0 - \beta_1 y \tag{2.8}$$

Then, substituting equation 2.8 into equation 2.7 gives the relationship between e and y as:

$$e = \beta_0 y - \beta_1 y^2 \tag{2.9}$$

For β_1 sufficiently small in relation to β_0, the e/y relationship takes the form of an inverted U, as shown in Figure 2.8(b). With this form of relationship, economic growth means higher emissions per capita until per capita income reaches the turning point, and thereafter actually reduces emissions per capita.

It has been hypothesised that a relationship like that shown in Figure 2.8(b) holds for many forms of environmental degradation. Such a relationship is called an 'environmental Kuznets curve' (EKC) after Kuznets (1955), who hypothesised an inverted U for the relationship between a measure of inequality in the distribution of income and the level of income. If the EKC hypothesis held generally, it would imply that instead of being a threat to the environment as is often argued (see the discussion of *The Limits to Growth* below), economic growth is the means to environmental improvement. That is, as countries develop economically, moving from lower to higher levels of per capita income, overall levels of environmental degradation will eventually fall.

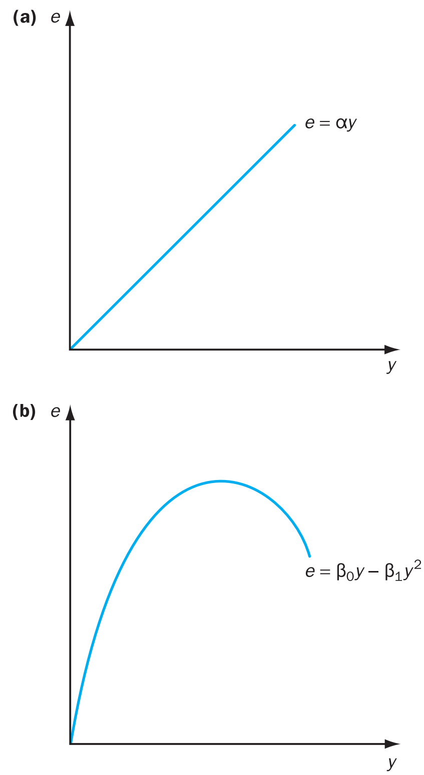


Figure 2.8 Environmental impact and income
Source: Adapted from Common (1996)

The argument for an EKC hypothesis has been succinctly put as follows:

> At low levels of development both the quantity and intensity of environmental degradation is limited to the impacts of subsistence economic activity on the resource base and to limited quantities of biodegradable wastes. As economic development accelerates with the intensification of agriculture and other resource extraction and the takeoff of industrialisation, the rates of resource depletion begin to exceed the rates of resource regeneration, and waste generation increases in quantity and toxicity. At higher

> levels of development, structural change towards information-intensive industries and services, coupled with increased environmental awareness, enforcement of environmental regulations, better technology and higher environmental expenditures, result in levelling off and gradual decline of environmental degradation.
>
> *Panayotou (1993)*

Clearly, the empirical status of the EKC hypothesis is a matter of great importance. If economic growth is actually and generally good for the environment, then it would seem that there is no need to curtail growth in the world economy in order to protect the global environment. In recent years there have been a number of studies using econometric techniques to test the EKC hypothesis against the data. Some of the results arising are discussed below. According to one economist, the results support the conclusion that

> there is clear evidence that, although economic growth usually leads to environmental degradation in the early stages of the process, in the end the best – and probably the only – way to attain a decent environment in most countries is to become rich.
>
> *Beckerman (1992)*

Assessing the validity of this conclusion involves two questions. First, are the data generally consistent with the EKC hypothesis? Second, if the EKC hypothesis holds, does the implication that growth is good for the global environment follow? We now consider each of these questions.

2.2.5.2.1 Evidence on the EKC hypothesis

In one of the earliest empirical studies, Shafik and Bandyopadhyay (1992) estimated the coefficients of relationships between environmental degradation and per capita income for ten different environmental indicators as part of a background study for the *World Development Report 1992* (World Bank, 1992). The indicators are lack of clean water, lack of urban sanitation, ambient levels of suspended particulate matter in urban areas, urban concentrations of sulphur dioxide, change in forest area between 1961 and 1986, the annual rate of deforestation between 1961 and 1986, dissolved oxygen in rivers, faecal coliforms in rivers, municipal waste per capita, and carbon dioxide emissions per capita. Some of their results, in terms of the relationship fitted to the raw data, are shown in Figure 2.9. Lack of clean water

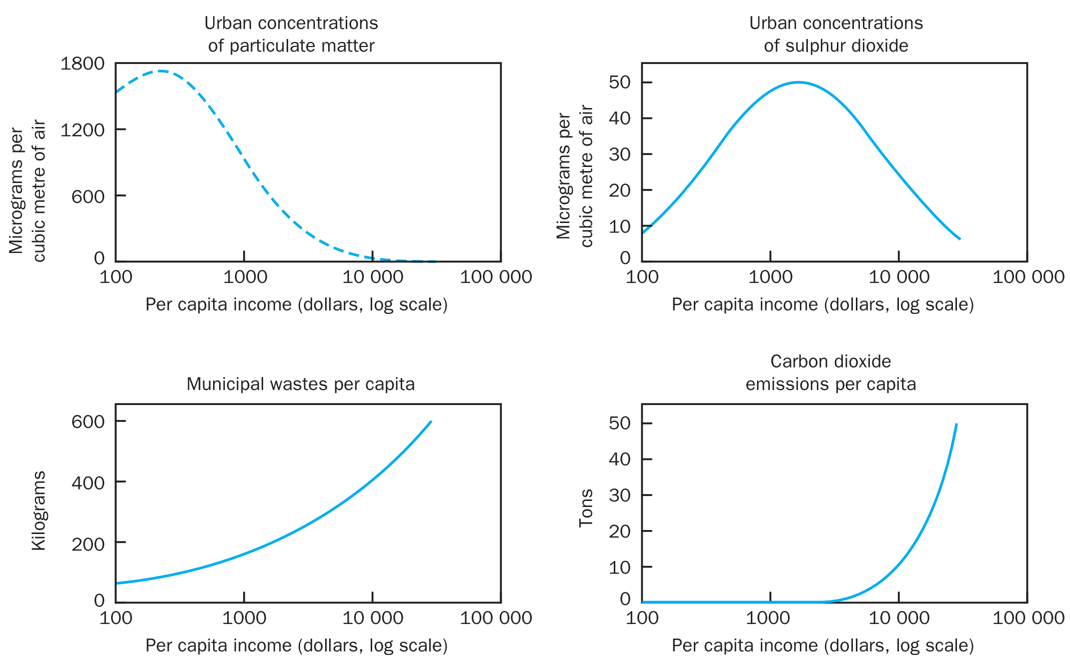


Figure 2.9 Some evidence from the EKC. Estimates are based on cross-country regression analysis of data from the 1980s
Source: Adapted from World Bank (1992)

and lack of urban sanitation were found to decline uniformly with increasing income. The two measures of deforestation were found not to depend on income. River quality tends to worsen with increasing income. As shown in Figure 2.9, two of the air pollutants were found to conform to the EKC hypothesis. Note, however, that CO_2 emissions, a major contributor to the 'greenhouse gases' to be discussed in relation to global climate change in Chapter 9, do not fit the EKC hypothesis, rising continuously with income, as do municipal wastes. Shafik and Bandyopadhyay summarise the implications of their results by stating:

> It is possible to 'grow out of' some environmental problems, but there is nothing automatic about doing so. Action tends to be taken where there are generalised local costs and substantial private and social benefits.

Panayotou (1993) investigated the EKC hypothesis for: sulphur dioxide (SO_2), nitrogen oxide (NO_x) suspended particulate matter (SPM) and deforestation. The three pollutants are measured in terms of emissions per capita on a national basis. Deforestation is measured as the mean annual rate of deforestation in the mid-1980s. All the fitted relationships are inverted Us, consistent with the EKC hypothesis. The result for SO_2 is shown in Figure 2.10, where the turning point is around \$3000 per capita.

There is now an extensive literature investigating the empirical status of the EKC hypothesis. The Further Reading section at the end of the chapter provides points of entry to this literature, and the key references. Some economists take the results in the literature as supporting the EKC for local and regional impacts, such as sulphur for example, but not for global impacts, such as carbon dioxide for example. However, Stern and Common (2001) present results that are not consistent with the existence of an EKC for sulphur. The EKC hypothesis may hold for some environmental impacts, but it does not hold for all.

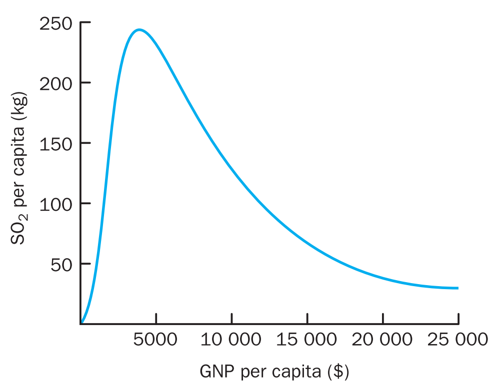


Figure 2.10 An EKC for SO_2
Source: Adapted from Panayotou (1993)

2.2.5.2.2 Implications of the EKC

If the EKC hypothesis were confirmed, what would it mean? Relationships such as that shown in Figure 2.10 might lead one to believe that, given likely future levels of income per capita, the global environmental impact concerned would decline in the medium-term future. In Figure 2.10 the turning point is near world mean income. In fact, because of the highly skewed distribution for per capita incomes, with many more countries – including some with very large populations – below rather than above the mean, this may not be what such a relationship implies.

This is explored by Stern *et al.* (1996), who also critically review the literature on the existence of meaningful EKC relationships. Stern *et al.* use the projections of world economic growth and world population growth published in the *World Development Report 1992* (World Bank, 1992), together with Panayotou's EKC estimates for deforestation and SO_2 emissions, to produce global projections of these variables for the period 1990–2025. These are important cases from a sustainable development perspective. SO_2 emissions are a factor in the acid rain problem: deforestation, especially in the tropics, is considered a major source of biodiversity loss. Stern *et al.* projected population and economic growth for every country in the world with a population greater than 1 million in 1990. The aggregated projections give world population growing from 5265 million in 1990 to 8322 million in 2025, and mean world per capita income rising from \$3957 in 1990 to \$7127 in 2025. They then forecast deforestation and SO_2 emissions for each country individually using the coefficients estimated by Panayotou. These forecasts were aggregated to give global projections for forest cover and SO_2 emissions.

Notwithstanding the EKC relationship shown in Figure 2.10, total global SO_2 emissions rise from 383 million tonnes in 1990 to 1181 million tonnes in 2025; emissions of SO_2 per capita rise from 73 kg to 142 kg from 1990 to 2025. Forest cover declines from 40.4 million km^2 in 1990 to a minimum of 37.2 million km^2 in 2016, and then increases to 37.6 million km^2 in 2025. Biodiversity loss on account of deforestation is an irreversible environmental impact, except on evolutionary time-scales, so that even in this case the implications of the fitted EKC are not reassuring.

Generally, the work of Stern *et al*. shows that the answer to the second question is that even if the data appear to confirm that the EKC fits the experience of individual countries, it does not follow that further growth is good for the global environment. Arrow *et al*. (1995) reach a similar position on the relevance of the EKC hypothesis for policy in relation to sustainability. They note that:

> The general proposition that economic growth is good for the environment has been justified by the claim that there exists an empirical relation between per capita income and some measures of environmental quality.

They then note that the EKC relationship has been 'shown to apply to a selected set of pollutants only', but that some economists 'have conjectured that the curve applies to environmental quality generally'. Arrow *et al*. conclude that

> Economic growth is not a panacea for environmental quality; indeed it is not even the main issue

and that

> policies that promote gross national product growth are not substitutes for environmental policy.

In Box 2.3 we report some simulation results that indicate that even if an EKC relationship between income and environmental impact is generally applicable, given continuing exponential income growth, it is only in very special circumstances that there will not, in the long run, be a positive relationship between income and environmental impact.

Box 2.3 The environmental Kuznets curve and environmental impacts in the very long run

The environmental Kuznets curve (EKC) implies that the magnitude of environmental impacts of economic activity will fall as income rises above some threshold level, when both these variables are measured in per capita terms. Here we assume for the sake of argument that the EKC hypothesis is correct. Common (1995) examines the implications of the EKC hypothesis for the long-run relationship between environmental impact and income. To do this he examines two special cases of the EKC, shown in Figure 2.11.

In case **a** environmental impacts per unit of income eventually fall to zero as the level of income rises. Case **b** is characterised by environmental impacts per unit income falling to some minimum level, k, at a high level of income, and thereafter remaining constant at that level as income continues to increase. Both of these cases embody the basic principle of the EKC, the only difference being whether environmental impacts per unit income fall to zero or just to some (low) minimum level.

Suppose that the world consists of two countries that we call 'developed' and 'developing' which are growing at the same constant rate of growth, g. However, the growth process began at an earlier date in the developed country and so at any point in time its per capita income level is higher than in the developing country. Common investigates what would

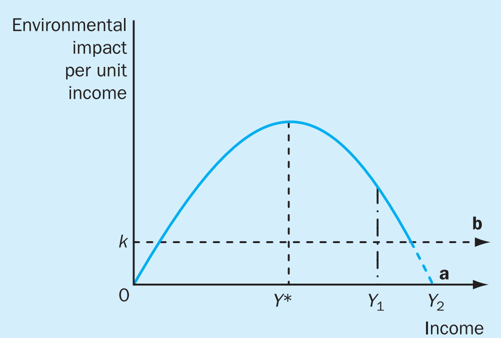


Figure 2.11 Two possible shapes of the environmental Kuznets curve in the very long run
Source: Adapted from Common (1995)

Box 2.3 *continued*

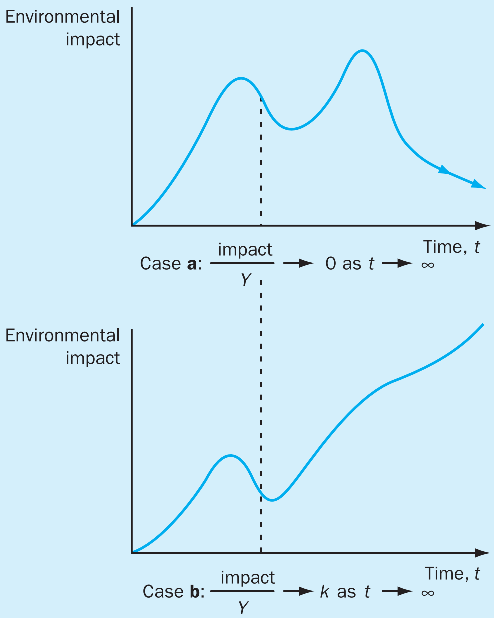


Figure 2.12 Two scenarios for the time profile of environmental impacts
Source: Adapted from Common (1995)

happen in the long run if case **a**, the highly optimistic version of the EKC, is true. He demonstrates that the time path of environmental impacts one would observe would be similar to that shown in the upper part of Figure 2.12. Why should there be a dip in the central part of the curve? For some period of time, income levels in the two countries will be such that the developed country is on the downward-sloping portion of its EKC while the developing country is still on the upward-sloping part of its EKC. However, as time passes and growth continues, both countries will be at income levels where the EKC curves have a negative slope; together with the assumption in case **a** that impacts per unit income fall to zero, this implies that the total level of impacts will itself converge to zero as time becomes increasingly large.

But now consider case **b**. No matter how large income becomes the ratio of environmental impacts to income can never fall below some fixed level, k. Of course, k may be large or small, but this is not critical to the argument at this point; what matters is that k is some constant positive number. As time passes, and both countries reach high income levels, the average of the impacts-to-income ratio for the two countries must converge on that constant value, k. However, since we are assuming that each country is growing at a fixed rate, g, the total level of impacts (as opposed to impacts per unit income) must itself eventually be increasing over time at the rate g. This is shown in the lower part of Figure 2.12.

What is interesting about this story is that we obtain two paths over time of environmental impacts which are entirely different from one another in qualitative terms for very small differences in initial assumptions. In case a, k is in effect zero, whereas in case **b**, k is greater than zero. Even if environmental impacts per unit of income eventually fell to a tiny level, the total level of impacts would eventually grow in line with income.

Which of these two possibilities – case **a** or case **b** – is the more plausible? Common argues that the laws of thermodynamics imply that k must be greater than zero. If so, the very-long-run relationship between total environmental impacts and the level of world income would be of the linear form shown (for per capita income) in panel (a) of Figure 2.8. The inference from the inverted U shape of the EKC that growth will reduce environmental damage in the very long run would be incorrect.

2.3 Poverty and inequality

Each year the United Nations Development Programme (UNDP) produces a *Human Development Report*, which draws on reports and data collections from a wide range of United Nations and other international agencies, and is the most useful single source of data and analysis on the current global state of humanity. This section draws heavily on these reports.

In the *Human Development Report 2001* (UNDP, 2001) it is stated that:

> Human development challenges remain large in the new millennium . . . Across the world we see

> unacceptable levels of deprivation in people's lives. Of the 4.6 billion people in developing countries, more than 850 million are illiterate, nearly a billion lack access to improved water sources, and 2.4 billion lack access to basic sanitation. Nearly 325 million boys and girls are out of school. And 11 million children under age five die each year from preventable causes – equivalent to more than 30 000 a day. Around 1.2 billion people live on less than $1 a day (1993 PPP US$), and 2.8 billion live on less than $2 a day.

In its report for 1998 (UNDP, 1998), the UNDP had additionally noted that of the population of the developing nations: 1.1 billion lacked adequate housing; 0.9 billion were undernourished; 0.9 billion had no access to modern health services.

Against this background, the report for 2001 comments that:

> The magnitude of these challenges appears daunting. Yet too few people recognize that the impressive gains in the developing world in the last 30 years demonstrate the possibility of eradicating poverty. A child born today can expect to live eight years longer than one born 30 years ago. Many more people can read and write, with the adult literacy rate having increased from an estimated 47% in 1970 to 73% in 1999. The share of rural families with access to safe water has grown more than fivefold. Many more people can enjoy a decent standard of living, with average incomes in developing countries having almost doubled in real terms between 1975 and 1998, from $1300 to $2500 (1985 PPP US$).

We now examine the current situation and recent trends in a little more detail. For a fuller version of what is a complex story, the reader should consult recent editions of the *Human Development Report*, and other references provided in the Further Reading section at the end of the chapter.

2.3.1 The current state of human development

Table 2.5 gives data on a number of indicators taken from the *Human Development Report 2007/2008*. The data cover 177 nations. There are 17 members of the UN that are not included in these data on the grounds that reliable information for them is not available. The excluded nations have an aggregate population of about 100 million out of a total world population of about 6.5 billion. The largest excluded nations are: Afghanistan, Democratic Republic of Korea, Iraq, Serbia and Somalia.

The nations of the world are grouped in different ways in different contexts. The three groupings in Table 2.5 are one of the classifications used in the *Human Development Report*. OECD stands for 'Organisation for Economic Co-operation and Development'. This organisation has 30 members, and corresponds roughly to the set of advanced industrial nations sometimes referred to as the 'developed world' or 'the North'. As indicated, Turkey is a member, as is Mexico. 'CE, EE and CIS'

Table 2.5 International comparisons at the start of the twenty-first century

	Life expectancy[a]	Infant mortality[b]	% of population undernourished[c]	GDP per capita[d]	Electricity per capita[e]
World	68.1	52	17	9 543	2 701
OECD	78.3	9	..	29 197	8 795
USA	77.9	6	<2.5	41 890	14 240
Turkey	71.4	26	3	8 407	2 122
CE, EE and CIS	68.6	22	..	9 527	4 539
Hungary	72.9	7	<2.5	17 887	4 070
Uzbekistan	66.8	57	25	2 063	1 944
Developing	66.1	57	17	5 282	1 221
Least developed	54.5	97	35	1 499	119
Sub-Saharan	49.6	102	32	1 998	478

[a] Years at birth, 2005, Table 1, UNDP (2007)
[b] Per 1000 live births, 2005, Table 10, UNDP (2007)
[c] 2002/4, Table 7, UNDP (2007) (.. means not available)
[d] 2005, PPP US$, Table 1, UNDP (2007)
[e] Kilowatt hours, 2004, Table 22, UNDP (2007)

is short for 'Central and Eastern Europe and the Commonwealth of Independent States', which is the former Soviet Union and its satellites. This grouping includes countries at very different levels of development, as illustrated by Hungary and Uzbekistan. All of the nations that are covered by the data but in neither the OECD nor CE, EE and CIS are classified as 'developing'. For most indicators the *Human Development Report* provides data for several subsets of this classification, two of which are included in Table 2.5. Many of the 'least developed' nations are located in Sub-Saharan Africa; non-African members of the least developed nations set include Bangladesh, Cambodia, Haiti, Lao People's Democratic Republic, Myanmar and Nepal.

The 2005 population sizes for the three groups of nations are: OECD 1173 million; CE, EE and CIS 405 million; developing 5215 million. The population of the set of least developed is 766 million, and for Sub-Saharan Africa it is 723 million.

On average, people in the OECD can expect to live almost 12 years longer than people in the developing world. OECD infant mortality is less than one-fifth the rate in the developing world. Undernourishment is rare in most of the countries of the OECD and CE, EE and CIS: <2.5 means less than 2.5%. Almost one-fifth of the world's total population are undernourished, and almost one-third of those in the worse-off parts of the developing world are. In round-number terms, the *Human Development Report* follows the World Bank and defines poverty as an annual income of less than PPP US$600 in terms of current PPP $s – this corresponds to $1 per day in terms of 1993 PPP US$. On that basis, according to Table 2.5, even for the least developed nations average income is above the poverty line. However, as quoted above, looking behind the average, it is estimated that 1.2 billion people are below the poverty line. In terms of averages, GDP per capita in the OECD is more than five times that in the developing world, and more than 19 times that in the least developed nations. For electricity consumption per capita, the relativities are broadly the same as for income per capita.

The data in Table 2.5 are for just a small sample of the possible indicators of human development. The picture that they show is broadly the same across all indicators – many human beings currently experience poverty and deprivation, and there are massive inequalities. In regard to income inequality, the *Human Development Report 2001* (UNDP, 2001) cites some results from a study that is based on household survey data rather than national income data. The study relates to the period 1988–1993 and covers 91 countries with 84% of world population. According to this study:

- The income of the poorest 10% was 1.6% of that of the richest 10%.
- The richest 1% of the world population received as much income in total as the poorest 57%.
- Around 25% of the population received 75% of total income.

2.3.2 Recent trends

An important question is whether things have been getting better in recent history. Table 2.6 shows the ratio of the values taken by the Table 2.5 indicators to their values as near to a quarter of a century ago as the data allow.

Life expectancy increased proportionately more in the developing world than in the OECD. It actually decreased for the CE, EE and CIS as a whole, though in some of its constituent nations it did increase a little. In the Russian Federation, life expectancy decreased from 69 for 1970–1975 to 64.8 for 2000–2005. This is associated with economic collapse and a major breakdown in preventive health care. Also culpable may be the cumulative effects of serious environmental contamination over many years in the Soviet Union, especially toxic wastes from chemical plants, pesticides from agriculture and nuclear radiation from various sources. For infant mortality a ratio of less than one indicates improvement over the period. Considering the three groupings, the improvement was least in CE, EE and CIS. Looking at undernourishment, there are no data for the OECD and CE, EE and CIS. It is known that for some countries in the latter it increased, as shown for Uzbekistan in Table 2.6. For the developing world as a whole and for the sub-groups shown in Table 2.6, the proportion of the population undernourished fell. Given the increase in population in developing countries, the absolute numbers undernourished fell by less than is shown in Table 2.6.

Table 2.6 Ratios for recent change

	Life expectancy[a]	Infant mortality[b]	% of population undernourished[c]	GDP per capita[d]	Electricity per capita[e]
OECD	1.11	0.22	..	1.81	1.42
USA	1.08	0.30	..	1.81	1.33
Turkey	1.24	0.17	..	1.71	3.08
CE, EE and CIS	0.99	0.56	..	1.52	..
Hungary	1.04	0.19	..	1.47	1.21
Uzbekistan	1.05	0.69	3.13	0.89	..
Developing	1.17	0.52	0.81	2.10	2.40
Least Developed	1.18	0.64	0.92	1.31	1.29
Sub-Saharan	1.07	0.71	0.89	0.86	1.04

[a] For 2000–05 divided by 1970–75, Table 10, UNDP (2007)
[b] 2005 divided by 1970, Table 8, UNDP (2007)
[c] 2002/4 divided by 1990/2, Table 7, UNDP (2007) (.. means not available)
[d] 2005 divided by 1975, calculated from annual growth rates, Table 14, UNDP (2007)
[e] 2004, from Table 22, UNDP (2007), divided by 1980 from Table 19 UNDP (2002)

In the developing world as a whole GDP per capita grew by more than it did in the OECD, more than doubling. However, here again, there is much variation within the developing world. For the least developed nations, per capita income increased by just 31% over 30 years, and for Sub-Saharan Africa it actually fell by 14%. There was also a lot of variation within the CE, EE and CIS group, where, as shown for Uzbekistan, GDP per capita fell in some countries. Over the 1990s the total number of people living below the poverty line as defined above was more or less constant at 1.2 billion. Given that the world population grew over this period, the proportion of the world's population living in poverty so defined fell slightly.

What about inequality? The *Human Development Report 2001* (Box 2.3) reports calculations based on GDP per capita data which show that from 1970 to 1997 the ratio of the income of the richest 10% to the poorest 10% increased from 19.4 to 26.9, indicating increasing inequality. On the other hand, if the ratio is calculated for the top and bottom 20% it falls from 14.9 to 13.1, indicating decreasing inequality.

Table 2.7 is based on the GDP per capita data used for Tables 2.5 and 2.6. It shows the ratio of GDP per capita for a group or nation to that of the USA for the same year. Thus, for the OECD as a whole GDP per capita was 70% of that of the USA in 1975 and 2005 – the OECD and the USA grew at the same rate, so that relative to the latter the former became neither better off nor worse off – the inequality remained constant. Turkey grew faster than the USA and the degree of inequality was reduced. For CE, EE and CIS as a whole, and for Uzbekistan and Hungary the ratio fell so that inequality increased. For the developing world as a whole, inequality in relation to the USA decreased a little, in that the ratio increases from 0.11 to 0.13. However, for the least developed nations and Sub-Saharan Africa, in relation to the USA and – given the above observations on the USA and the OECD – the OECD as a whole, income inequality increased. In the case of Sub-Saharan Africa, per capita GDP fell by 14% while USA per capita GDP increased by about 80% (Table 2.6), and the ratio of the former to the latter fell by 50%.

Table 2.7 GDP per capita relativities to the USA

	1975	2005
OECD	0.70	0.70
USA	1.00	1.00
Turkey	0.12	0.20
CE, EE and CIS	0.27	0.23
Hungary	0.53	0.43
Uzbekistan	0.10	0.05
Developing	0.11	0.13
Least developed	0.05	0.04
Sub-Saharan	0.10	0.05

Calculated from Tables 2.5 and 2.6

2.3.3 Growth as the solution

Economists have a very strong attachment to economic growth as a major policy objective. A major reason for this is that they see it as the only feasible way to solve the problem of poverty. The argument is that with economic growth, the lot of the poor can be improved without taking anything away from the better-off. Generally the better-off will resist attempts to redistribute from them to the poor, so that this route to poverty alleviation will involve social tension and possibly violent conflict. Further, over and above such considerations, poverty alleviation via redistribution may not work even if it is politically and socially feasible. The problem is that typically the poor are much more numerous than the rich, so that there is simply not enough to take from the rich to raise the poor above the poverty line. When, in the years following the Second World War, economists thought that they understood how to bring about economic growth they came to think that they could solve an age-old problem of the human condition – they came to think that the poor need not always be with us.

Indeed, perhaps the most famous economist of the twentieth century, J.M. Keynes, saw in economic growth the prospect that the very problem that was taken to be the essential economic problem – scarcity – would be abolished, so that economists would become largely redundant. In an essay (Keynes, 1931), written in the early 1930s, on the economic prospects for the grandchildren of adults then alive, Keynes was concerned to put in perspective the waste entailed in the then-prevalent under-use of available resources, especially labour. If the means to avoid such waste could be found and adopted, Keynes argued, economic growth, i.e. increasing per capita GDP, at 2% per year would easily be attained and sustained. This, he pointed out, would mean that in one hundred years output would increase sevenfold. Scarcity would be abolished, and a situation arise in which economics and economists were no longer important. In the years after the Second World War, most economists thought that Keynesian macroeconomics was the means to achieving full employment and sustained growth throughout the world.

The arithmetic of compound growth – growth at a constant proportional rate – is indeed striking. And, there is no doubt that historically economic growth has raised the consumption levels of the mass of the population in the rich industrial world to levels that could scarcely have been conceived of at the start of the industrial revolution, 200 years ago. There is also no doubt that for the developing world as a whole, economic growth in the latter part of the twentieth century reduced the extent of poverty. The arithmetic of economic growth does not, however, necessarily imply any reduction in economic inequality. If the incomes of the rich and the poor grow at the same rates, the proportionate difference between them stays the same, and the absolute difference – in dollars per year – actually increases. The original Kuznets curve hypothesis was that, with growth, income inequality first increased then decreased. The evidence on this hypothesis is mixed. As noted above, global income inequalities have not generally decreased in recent years. Within some advanced economies inequality has increased.

2.4 Limits to growth?

An important event in the emergence in the last three decades of the perception that there is a sustainability problem was the publication in 1972 of a book, *The Limits to Growth* (Meadows *et al*., 1972), which was widely understood to claim that environmental limits would cause the collapse of the world economic system in the middle of the twenty-first century.

The book was roundly condemned by most economists, but influenced many other people. It is arguable that it was a stimulus to the re-emergence of interest in natural resources on the part of economists in the early 1970s noted in Chapter 1. One economist argued, at around the same time, that the limits to growth were social rather than environmental.

2.4.1 Environmental limits

The Limits to Growth reported the results of a study in which a computer model of the world system, World3, was used to simulate its future. World3 represented the world economy as a single economy, and included interconnections between that economy and its environment. According to its creators, World3

> was built to investigate five major trends of global concern – accelerating industrialization, rapid population growth, widespread malnutrition, depletion of non-renewable resources, and a deteriorating environment. These trends are all interconnected in many ways, and their development is measured in decades or centuries, rather than in months or years. With the model we are seeking to understand the causes of these trends, their interrelationships, and their implications as much as one hundred years in the future.
>
> *Meadows et al. (1972), p. 21*

It incorporated:

(a) a limit to the amount of land available for agriculture;
(b) a limit to the amount of agricultural output producible per unit of land in use;
(c) a limit to the amounts of non-renewable resources available for extraction;
(d) a limit to the ability of the environment to assimilate wastes arising in production and consumption, which limit falls as the level of pollution increases.

The behaviour of the economic system was represented as a continuation of past trends in key variables, subject to those trends being influenced by the relationships between the variables represented in the model. These relationships were represented in terms of positive and negative feedback effects. Thus, for example, population growth is determined by birth- and death-rates, which are determined by fertility and mortality, which are in turn influenced by such variables as industrial output per capita, the extent of family planning and education – for fertility – and food availability per capita, industrial output per capita, pollution, and the availability of health care – for mortality. The behaviour over time, in the model of each of these variables, depends in turn on that of others, and affects that of others.

On the basis of a number of simulations using World3, the conclusions reached by the modelling team were as follows:

> 1. If the present growth trends in world population, industrialization, pollution, food production and resource depletion continue unchanged, the limits to growth on this planet will be reached sometime within the next 100 years. The most probable result will be a sudden and uncontrollable decline in both population and industrial capacity.
> 2. It is possible to alter these trends and to establish a condition of ecological and economic stability that is sustainable far into the future. The state of global equilibrium could be designed so that the basic material needs of each person on earth are satisfied and each person has an equal opportunity to realize his or her individual human potential.
> 3. If the world's people decide to strive for this second outcome rather than the first, the sooner they begin working to attain it, the greater will be their chances of success.
>
> *Meadows et al. (1992), p. xiii*

What *The Limits to Growth* actually said was widely misrepresented. It was widely reported that it was an unconditional forecast of disaster sometime in the next century, consequent upon the world running out of non-renewable resources. In fact, as the quotation above indicates, what was involved was conditional upon the continuation of some existing trends. Further, this conditional prediction was not based upon running out of resources.

The first model reported run did show collapse as the consequence of resource depletion. Figure 2.13 is a reproduction of the figure in *The Limits to Growth* that reports the results for 'World Model Standard Run'. This run assumes no major changes in social, economic or physical relationships. Variables follow actual historical values until the year 1970. Thereafter, food, industrial output and population grow exponentially until the rapidly diminishing resource base causes a slowdown in industrial growth. System lags result in pollution and population continuing to grow for some time after industrial output has peaked. Population growth is finally halted by a rise in the mortality rate, as a result of reduced flows of food and medical services.

However, the next reported run involved the model modified by an increase in the resource availability limit such that depletion did not give rise to problems for the economic system. In this run, the proximate source of disaster was the level of pollution consequent upon the exploitation of the increased amount of resources available, following from the materials balance principle. A number of variant model runs were reported, each relaxing some constraint. The conclusions reached were

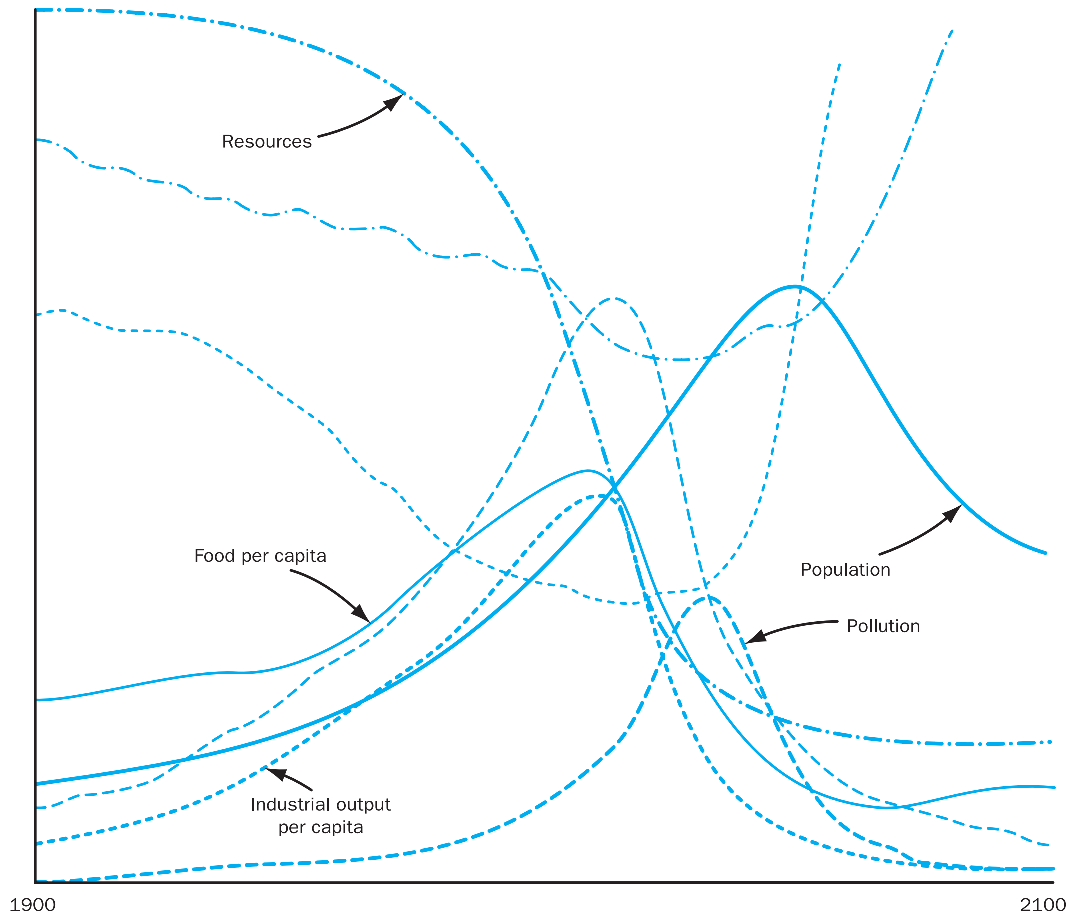


Figure 2.13 Base run projections of the 'limits to growth' model
Source: Meadows *et al*. (1972), page 124

based on consideration of all of the variant model runs. Successive runs of the model were used to ascertain those changes to the standard configuration that were necessary to get the model to a sustainable state, rather than to collapse mode.

It was widely reported that the World3 results said that there were limits to 'economic growth'. In fact, what they said, as the conclusions quoted above indicate, is that there were limits to the growth of material throughput for the world economic system. As economic growth is measured it includes the consumption of the output of the service sector, as well as the agricultural and industrial sectors.

A sequel (Meadows *et al*., 1992) to *The Limits to Growth*, written by the same team and entitled *Beyond the Limits*, was published in 1992 to coincide with the UNCED conference held in Rio de Janeiro. The publication of the sequel generated much less controversy than the original did. This might suggest some major change in analysis and conclusions as between original and sequel. In fact there is very little substantive difference in the conclusions, and apart from updating of numerical values used, the model is stated to be modified in only minor ways from the original World3. The position on this as stated in the sequel is:

> As far as we can tell from the global data, from the World3 model, and from all we have learned in the past twenty years, the three conclusions we drew in *The Limits to Growth* are still valid, but they need to be strengthened.
>
> *Meadows et al. (1992), p. xv*

The quotation at the beginning of this chapter is from a second sequel to *The Limits to Growth*, with the title *Limits to Growth: The 30-Year Update* and published in 2005 (Meadows *et al*., 2005). As compared to the first sequel, this book uses a slightly modified World3, presents similar model-generated scenarios, and reaches similar conclusions. The authors state that 'they are much more pessimistic about the global future' than they were in 1972, and that:

> It is a sad fact that humanity has largely squandered the past 30 years in futile debates and well-intentioned, but halfhearted, responses to the global ecological challenge. We do not have another 30 years to dither. Much will have to change if the ongoing overshoot is not to be followed by collapse during the twenty-first century.
>
> *Meadows et al. (2005), p. xvi*

2.4.2 Economists on environmental limits

The response by economists to *The Limits to Growth* was almost entirely hostile. Given their commitment to economic growth as the solution to the problem of poverty and the widespread existence of the problem, noted in the previous section, this was hardly surprising. Prominent among the critical responses from economists were those by Page (1973), Nordhaus (1972), Beckerman (1972, 1974), Cole *et al*. (1973) and Lecomber (1975). According to one eminent economist it was 'a brazen, impudent piece of nonsense that nobody could possibly take seriously' (Beckerman, 1972). As noted above, economists have had much less to say, and much less critical things to say, about the sequel, *Beyond the Limits*. In a foreword to it, a Nobel laureate in economics, Jan Tinbergen, says of it: 'We can all learn something from this book, especially we economists.'

The main line of the criticism of the original by economists was that the feedback loops in World3 were poorly specified in that they failed to take account of behavioural adjustments operating through the price mechanism. In particular, it was argued that changing patterns of relative scarcity would alter the structure of prices, inducing behavioural changes in resource-use patterns. Given a well-functioning market mechanism, it was argued, limits to growth would not operate in the way reported by the modelling team. It was conceded by some of the economist critics that the force of this argument was weakened by the fact that for many environmental resources and services, markets did not exist, or functioned badly where they did. However, it was also argued that such 'market failure' could be corrected by the proper policy responses to emerging problems. This presumes that the sorts of substitutions for environmental services that we discussed above can be made, given properly functioning markets or policy-created surrogates for such, to the extent that will overcome limits that would otherwise exist. A major, and largely unresolved, question in the debates about the existence of a sustainability problem is the existence and effectiveness of substitutes for environmental services.

2.4.3 Social limits to growth

The argument for 'social limits to growth' was first advanced in a book with that title (Hirsch, 1977), published five years after *The Limits to Growth*. Hirsch argued that the process of economic growth becomes increasingly unable to yield the satisfaction which individuals expect from it, once the general level of material affluence has satisfied the main biological needs for life-sustaining food, shelter and clothing. As the average level of consumption rises, an increasing portion of consumption takes on a social as well as an individual aspect, so that

> the satisfaction that individuals derive from goods and services depends in increasing measure not only on their own consumption but on consumption by others as well.
>
> *Hirsch (1977), p. 2*

The satisfaction a person gets from the use of a car, for example, depends on how many other people do the same. The greater the number of others who use cars, the greater is the amount of air pollution and the extent of congestion, and so the lower is the satisfaction one individual's car use will yield. However, Hirsch's main focus was on what he calls 'positional goods', the satisfaction from which depends upon the individual's consumption relative to that of others, rather than the absolute level of consumption. Consider, as an example, expenditure on education in an attempt to raise one's chances of securing sought-after jobs. The utility to a person of a given level of educational expenditure will decline as an increasing number of others also attain that level of education. Each person purchasing education seeks to gain individual advantage, but the simultaneous actions of others frustrate these objectives for each individual. As the average level of education rises, individuals will not receive the gains they expect from higher qualifications.

Once basic material needs are satisfied, further economic growth is associated with an increasing proportion of income being spent on such positional goods. As a consequence, Hirsch argued, growth in developed economies (such as members of the OECD) is a much less socially desirable objective than economists have usually thought. It does not deliver the increased personal satisfactions that it is supposed to. Traditional utilitarian conceptions of social welfare (see Chapters 3 and 4) may be misleading in such circumstances, as utilities are interdependent. Using terminology to be introduced and explained in Chapter 4, we can say that given the external effects arising due to the consumption of others affecting the utility that an individual derives from his or her own consumption, the simple summation of individual consumption levels overstates collective welfare.

Since the 1950s social scientists have been conducting surveys in which they ask individuals about how satisfied with their lives, or happy, they are. There is now a substantial body of data that can, among other things, be used to study the relationship between economic growth and happiness. As will be explained in Section 3.3.4 of the next chapter, the evidence from these data are consistent with Hirsch's argument. If, for example, we look at a plot of happiness against per capita national income across countries at a point in time, as in Figure 3.4, we find that happiness increases with income, but at a decreasing rate. Across rich countries, in terms of the national average, additional per capita national income delivers little in terms of self-assessed happiness.

2.5 The pursuit of sustainable development

Social limits to growth are not currently a problem in developing countries. Many people now live in conditions such that basic material needs are not satisfied. This is particularly true for people living in the poor nations of the world, but is by no means restricted to them. Even in the richest countries, income and wealth inequalities are such that many people live in conditions of material and social deprivation. For many years, it was thought that the eradication of poverty required well-designed development programmes that were largely independent of considerations relating to the natural environment. The goal of economic and political debate was to identify growth processes that could allow continually rising living standards. Economic development and 'nature conservation' were seen as quite distinct and separate problems. By some commentators, concern for the natural environment was seen as a rather selfish form of self-indulgence on the part of the better-off.

Perspectives have changed significantly since the 1970s. While the pursuit of economic growth and development continues, it is recognised that the maintenance of growth has an important environmental dimension. During the 1970s, a concern for sustainability began to appear on the international political agenda, most visibly in the proceedings of a series of international conferences. The common theme of these debates was the interrelationship between poverty, economic development and the state of the natural environment.

Perhaps the best-known statement of the sustainability problem derives from the 1987 report of the World Commission on Environment and Development, which set the agenda for much of the subsequent discussion of sustainability.

2.5.1 The World Commission on Environment and Development

The World Commission on Environment and Development, WCED, was established in 1983 by the United Nations. Its mandate was:

1. to re-examine the critical environment and development issues and to formulate realistic proposals for dealing with them;
2. to propose new forms of international cooperation on these issues that will influence policies and events in the direction of needed changes;
3. to raise the levels of understanding and commitment to action of individuals, voluntary organisations, businesses, institutes and governments.

WCED comprised 23 commissioners from 21 different countries. The chairperson was Gro Harlem Brundtland, who had previously been both Minister for the Environment and Prime Minister of Norway. Over a period of two years, the commissioners held public meetings in eight countries, at which people could submit their views on WCED's work. In regard to analysis and awareness-raising, WCED focused on population growth, food security, biodiversity loss, energy, resource depletion and pollution, and urbanisation.

2.5.1.1 The Brundtland Report

The report that WCED produced in 1987 – *Our Common Future* (WCED, 1987) – is often referred to as 'the Brundtland report' after the name of the WCED chairperson. It advanced, with great effect, the concept of 'sustainable development', which is now on political agendas, at least at the level of rhetoric, around the world. The Brundtland report was, in political terms, an outstanding and influential piece of work.

It provides much information about what we have called here the sustainability problem, setting out the nature of economy–environment interdependence, identifying a number of potential environmental constraints on future economic growth, and arguing that current trends cannot be continued far into the future. Thus, according to the Brundtland report,

> Environment and development are not separate challenges: they are inexorably linked. Development cannot subsist on a deteriorating environmental base; the environment cannot be protected when growth leaves out of account the costs of environmental protection.
>
> *p. 37*

while

> The next few decades are crucial. The time has come to break out of past patterns. Attempts to maintain social and ecological stability through old approaches to development and environmental protection will increase instability.
>
> *p. 22*

The Brundtland report does not conclude that future economic growth is either infeasible or undesirable. Having defined sustainable development as development that

> seeks to meet the needs and aspirations of the present without compromising the ability to meet those of the future
>
> *p. 43*

it states that:

> Far from requiring the cessation of economic growth, it [sustainable development] recognizes that the problems of poverty and underdevelopment cannot be solved unless we have a new era of growth in which developing countries play a large role and reap large benefits.
>
> *p. 40*

Nor does it require that those nations already developed cease to pursue economic growth:

> Growth must be revived in developing countries because that is where the links between economic growth, the alleviation of poverty, and environmental conditions operate most directly. Yet developing countries are part of an interdependent world economy; their prospects also depend on the levels and patterns of growth in industrialized nations. The medium term prospects for industrial countries are for growth of 3–4 per cent, the minimum that international financial institutions consider necessary if these countries are going to play a part in expanding the world economy. Such growth rates could be environmentally sustainable if industrialized nations can continue the recent shifts in the content of their growth towards less material- and energy-intensive activities and the improvement of their efficiency in using materials and energy.
>
> *p. 51*

In the light of an appreciation of the economy–environment interdependence and the current level of global economic activity, some environmentalists have expressed the view that 'sustainable development' is an oxymoron. It is their assessment that the current situation is such that we already are at the limits of what the environment can tolerate, so that growth will inevitably damage the environment, and cannot, therefore, be sustainable. It is the assessment of the Brundtland report that environmental limits to growth can be avoided, given the adoption, worldwide, of policies to affect the form that economic growth takes. To make growth sustainable, those policies would have to involve reducing, at the global level, the material content of economic activity, economising in the use of resources as the value of output increases, and substituting the services of reproducible capital for the services of natural capital. Much of resource and environmental economics is about the policy instruments for doing that, as we shall see in later chapters.

Given the nature of the WCED, it is not surprising that the Brundtland report is not strong on detailed and specific policy proposals that would facilitate the move from 'past patterns' to sustainable development. It urges, for example, that national governments merge environmental and economic considerations in their decision making. It did make a specific recommendation regarding item 2 from its mandate. This was that the UN General Assembly convene an international conference

> to review progress made and promote follow-up arrangements that will be needed over time to set benchmarks and to maintain human progress within the guidelines of human needs and natural laws.
>
> *p. 343*

This recommendation was acted upon, and the result was the United Nations Conference on Environment and Development, UNCED, which took place in Rio de Janeiro in June 1992.

2.5.2 UNCED: Rio de Janeiro 1992

The conference itself was preceded by over two years of preparatory international negotiations. Delegations were sent from 178 nations and the meeting was attended by 107 heads of government (or state). During UNCED several parallel and related conferences took place in Rio de Janeiro; the meeting for 'non-governmental organisations', mainly pro-environment pressure groups, involved more participants than UNCED itself. It has been estimated that, in total, over 30 000 people went to Rio de Janeiro in June 1992.

The preparatory negotiations dealt with four main areas: draft conventions on biodiversity conservation, global climate change, forest management, and the preparation of two documents for adoption at UNCED. The main UNCED outcomes were as follows. There was complete agreement on the non-binding adoption of the *Rio Declaration* and *Agenda 21*. The first of these comprises 27 statements of principle in regard to global sustainable development. The second is an 800-page document covering over 100 specific programmes for the attainment of global sustainable development: many of these programmes involve resource transfers from the industrial to the developing nations. UNCED also agreed on the creation of a new UN agency, a Commission for Sustainable Development, to oversee the implementation of *Agenda 21*. Agreement was also reached on the, non-binding, adoption of a set of principles for forest management. The industrial nations reaffirmed their previous non-binding commitments to a target for development aid of 0.7% of their GNP. It should be noted that it is still true that only a few of the industrial nations actually attain this target.

Two conventions were adopted, by some 150 nations in each case, which would be binding on signatories when ratified by them. These covered global climate change and biodiversity conservation: the latter was not signed by the USA at the Rio meeting, but the USA did sign in 1993 after a change of administration. Although binding, these conventions did not commit individual nations to much in the way of specific actions. The *Convention on Biological Diversity* deals with two main issues – the exploitation of genetic material and biodiversity conservation. In regard to the latter, signatories agree to create systems of protected areas, for example, but undertake no commitments regarding their extent. The *Framework Convention on Climate Change* was mainly about the principles according to which future negotiations – known as Conferences of the Parties, COPS – were to try to establish commitments and rules. A major principle was that commitments would be limited to the developed nations.

Many environmental activists, as well as many concerned to promote economic development in poor nations, regarded the actual achievements at UNCED as disappointing, but it did confirm that sustainable development was, and would remain, firmly on the world political agenda. While specific commitments were not a major feature of the outcomes, there were agreements with the potential to lead to further developments. The creation of the Commission for Sustainable Development is clearly an important institutional innovation at the international level.

The convening of, and the outcomes at, UNCED suggest that the need to address the economic and environmental problems arising from economy–environment linkages is widely accepted. Equally, UNCED and subsequent events suggest that even when the existence of a problem is widely agreed by national governments, agreement on the nature of appropriate policy responses is limited. Further, there is clearly reluctance on the part of national governments to incur costs associated with policy responses, and agreed action is even more difficult to realise than agreement about what should be done. The difficulties involved in achieving international action on environmental problems, such as climate change, are discussed in Chapter 9, along with progress that has been made since 1987.

2.5.3 The UN and Sustainable Development after Rio de Janeiro

As noted, the 1992 Conference brought into being the UN Commission on Sustainable Development to oversee the UN's work on the pursuit of sustainable development. Information on the activities of the Commission can be accessed at its website, http://www.un.org/esa/sustdev/csd/review.htm. Within the UN's administration, the parallel body to the Commission is the Division for Sustainable Development, http://www.un.org/esa/sustdev/index.html, which is part of the Department of Economic and Social Affairs.

The Commission has organised a number of meetings to build on the 1992 Conference. The most notable of these was the World Summit on Sustainable Development, WSSD, held in September 2002 in Johannesburg, also often referred to as 'Rio + 10' for obvious reasons. WSSD was attended by 9000 government delegates, including 104 heads of government or state, 8000 NGO representatives, 4000 members of the press, and many representatives of business organisations. A full account of what happened, and what was agreed, at WSSD can be found via the website addresses given above. All nations signed up for the Johannesburg Plan of Implementation, which included limited (and mainly un-quantified) targets on such environmental issues as biodiversity loss, and a set of targets for economic development very similar to the Millennium Development Goals.

Most commentators were disappointed with the progress at WSSD beyond what had been achieved, or set in motion, at the 1992 conference. According to one:

> Compared to the 1992 Earth Summit in Rio, this summer's World Summit on Sustainable Development (WSSD) in Johnnesburg was bound to be somewhat disappointing. The negotiations leading up to Johannesburg had not provided any reason to expect dramatic break-throughs, and there were none. After the meeting, many non-governmental organisations denounced the WSSD as a failure. Even seasoned UN officials, while relieved that the Summit had not broken down completely, were rather muted in their responses.
>
> *Hilary French, Worldwatch Institute Policy Brief No 12, October 2002, downloaded April 2008 from http://www.worldwatch.org/node/1744.*

2.5.4 The Millennium Development Goals

Many would argue that while economic growth is necessary for dealing with poverty, it is not sufficient. As we have seen in this chapter, in the second half of the twentieth century growth in the world economy co-existed with the persistence of poverty. This led many to the view that dealing with poverty required a commitment to economic growth and policies directly aimed at poverty and other impediments to realising a good quality of life.

In September 2000 the UN convened the Millennium Summit at its headquarters in New York, which was attended by representatives of 189 UN members, most of whom were heads of government. The Summit adopted the Millennium Declaration, and its eight chapters were subsequently developed into an action plan with eight goals, which were called the Millennium Development Goals, MDGs. The plan calls for the MDGs to be achieved by 2015. Those goals are:

1. Eradicate extreme poverty and hunger.
2. Achieve universal primary education.
3. Promote gender equality and empower women.
4. Reduce child mortality.
5. Improve maternal health.
6. Combat HIV/AIDS, malaria, and other diseases.
7. Ensure environmental sustainability.
8. Develop a global partnership for development.

Each of these goals is related to more specific targets, of which there are 21 in all. In regard to the first goal, for example, the targets are to:

> halve, as compared with 1990, the proportion of people whose income is less than one dollar per day
> achieve full and productive employment and decent work for all, including women and young people
> halve, as compared with 1990, the proportion of people who suffer from hunger.

For more information on the MDGs and the targets visit the United Nations Development Programme (UNDP) website at http://www.undp.org/mdg/basics.shtml. Progress toward goals and targets is mixed. Looking, for example, at the income target noted above, for five of the 10 regions distinguished, the target is either 'not expected to be met by 2015' or there is 'no progress, or a deterioration or reversal', while for two 'target is already met or very close to being met' and for another two the target is expected to be met by 2015. For one region there is insufficient data. For the hunger target, the verdict is unfavourable for five regions, and favourable for five. The balance of favourable and unfavourable judgements of the prospects of success are similar across the other targets.[6]

Summary

Interdependence

The economy and the environment are linked through many complex relationships. Economic activity affects the environment, which affects the economy. The economy and the environment are interdependent systems.

Impact scale

The scale of humanity's current impact on the natural environment is without precedent, and raises the question of whether the natural environment can tolerate it without serious change, likely to be detrimental to human interests.

Impact drivers – IPAT

Humanity's impacts on the natural environment are driven by the size of the human population, the level of human affluence, and the technologies that humans use.

Behaviour

P, *A* and *T* all depend on human behaviour, which changes over time as the incentives facing people change. Policy affects the incentives that people face.

[6] These figures are taken from a chart prepared by the Statistics Division of the UN's Department of Economic and Social Affairs, and relate to 2008. The chart was downloaded from http://mdgs.un.org/unsd/mdg/default.aspx in January 2009. The corresponding figures for the same chart for 2007 were the same, as is true for most of the targets.

Poverty and inequality

The current human condition involves massive inequalities in many aspects of well-being and gross poverty for many millions of people.

Sustainable development

Sustainable development seeks to improve the lot of the world's poor without reducing the capacity of the natural environment to deliver a diverse set of services to the global economy.

Further reading

Data on the topics dealt with in this chapter can be found in the publications of agencies such as the United Nations (UN), the Food and Agriculture Organization of the United Nations (FAO), the United Nations Environment Programme (UNEP), the United Nations Development Programme (UNDP), the World Bank, the Organisation for Economic Co-operation and Development (OECD), the International Energy Agency (IEA), and the World Conservation Union (WCN, formerly the International Union for the Conservation of Nature, IUCN). Each year the World Resources Institute (WRI) collates data from these and other sources in a series of tables in its publication *World Resources*. This is jointly produced by WRI, UNEP, the World Bank and UNDP. Each year this publication also focuses on some particular aspect of economy–environment interdependence – for example, at the time of writing the latest available report was *World Resources 2000–2001: People and Ecosystems*. Some data are available from the websites of these organisations:

UN – www.un.org
FAO – www.fao.org/default.htm
UNEP – www.unep.org
UNDP – www.undp.org
World Bank – www.worldbank.org
OECD – www.oecd.org
IEA – www.iea.org
WRI – www.wri.org
WCN – www.iucn.org

Lomborg (2001) is a useful point of entry to the vast array of statistical materials on the current state of the environment and human development. The appearance of Lomborg's book in English generated a great deal of interest and controversy, as he argues that what the statistics show is that those 'environmentalists' who claim that the human condition is deteriorating and that the ability of the environment to support economic activity is decreasing are, overall, quite wrong. Many of those environmentalists have, in turn, claimed that Lomborg is both wrong and irresponsible. Lomborg treats each potential environmental problem in isolation, rather than as the set of linked phenomena that they are. Lomborg has a website, www.lomborg.com, which provides links to various contributions to the controversy.

There are a number of well-documented examples of unsustainable societies in human history, where collapse followed resource exhaustion. Ponting (1992) is an environmental history of humanity and provides brief accounts of some examples, and references to more detailed works; see also Diamond (1993) and Diamond (2006).

Jackson and Jackson (2000) and Park (2001) are two standard texts that deal at greater length with the environmental science topics covered in this chapter. Both are at an introductory level: Jackson and Jackson assumes some prior knowledge of chemistry. Krebs (2001) is a successful ecology text that is comprehensive but assumes no prior knowledge of the subject. Folke (1999) is a brief overview of ecological principles as they relate to ecological economics, and provides useful references to the literature. As it is set out here, the idea of resilience as a property of an ecosystem is developed in Holling (1986). The paper by Ludwig *et al*. (1997)

is a clear, but technical, exposition of the basic mathematics of Holling resilience and how it relates to another concept of resilience that appears in the ecology literature.

There is no uniquely correct way to classify the services that the environment provides to the economy. Barbier *et al*. (1994), in Table 3.1, provide a four-way classification of what they call the 'life support functions of ecosystems' into regulation, production, carrier and information functions. Costanza *et al*. (1997) distinguish 17 classes of 'ecosystem service'. Common (1995), Dasgupta (1982) and Perrings (1987) consider economy–environment interdependence and some of its implications from an economics perspective. D'Arge and Kogiku (1972) is an early contribution to the resource and environmental economics literature that contains a growth model which obeys the law of conservation of matter. As noted in the body of the chapter, the economist who did most to draw the attention of economists to the laws of thermodynamics and their implications for economics was Nicholas Georgescu-Roegen; various assessments of his contribution are presented in a special issue of the journal *Ecological Economics*, Vol. 22, no 3, September 1997.

Barbier *et al*. (1994) is a good introduction to biodiversity issues. Wilson's classic work (Wilson, 1988) on biodiversity has been updated as *Biodiversity II* (Reaka-Kudla *et al*., 1996). UNEP (1995) is a definitive reference work in this field, dealing primarily with definition and measurement of biodiversity loss, but also containing good chapters on economics and policy. See also Groombridge (1992) and Jeffries (1997) for excellent accounts of biodiversity from an ecological perspective. Measurement and estimation of biodiversity are examined in depth in Hawksworth (1995), and regular updated accounts are provided in the annual publication *World Resources*. The state of the worlds' ecosystems and biodiversity was extensively reviewed in the Millennium Ecosystem Assessment, information about which and publications arising from which can be accessed at http://www.millenniumassessment.org/. The extent of human domination of global ecosystems is considered in Vitousek *et al*. (1997); for the range of uncertainty attending such estimates see Field (2001). The Global Footprint Network, see http://www.footprintnetwork.org/, produces and reports annual update estimates of national and global ecological footprints. Wackernagel and Rees (1996) is a simple introduction to ecological footprints which describes methods of calculation and reports a variety of results. The organisation Redefining Progress is active in promoting the idea of ecological footprinting, and further information is available at www.redefiningprogress.org, where there can also be found work on Genuine Progress Indicators. In March 2000, *Ecological Economics*, Vol. 32, no 3, included a forum comprising 12 short papers on the ecological footprint. Several of these papers set out the limitations of the concept as a guide to policy.

The IPAT identity was introduced in Ehrlich and Holdren (1971); see also Ehrlich and Ehrlich (1990). It was originally set out as $I \equiv PCT$ where C stands for consumption, but IPAT is a better acronym than IPCT, and it is income per capita rather than consumption that matters and is most easily measured. The identity indicates the scale of technological change that is necessary to hold impact constant for any given change in population and/or affluence. The feasible prospects for technological change are discussed in von Weizşker *et al*. (1997) – it is claimed that T could be reduced by a factor of four, so that affluence could double and impact be cut by 50%, for constant population. Lovins *et al*. (2000) is even more optimistic about technological possibilities.

Becker (1960) is the classic original source of the literature on the economics of population. Easterlin (1978) provides a comprehensive and nonmathematical survey of the economic theory of fertility, and his 1980 volume provides an excellent collection of readings.

The EKC hypothesis was the subject of a special issue of the journal *Environment and Development Economics* in October 1997 (Vol. 2, part 4), and also of the journal *Ecological Economics* in May 1998 (Vol. 25, no 2). See also de Bruyn and Heintz (1999). Useful recent surveys and assessments are Dinda (2004), Stern (2004) and Müller-Fürstenberger and Wagner (2007).

The UNDP's annual *Human Development Report* is the best single source of data and commentary on the global situation in regard to affluence, poverty and inequality. As well as basic data, each year it reports country performance against a series of indices intended to capture several dimensions of human development. Arndt (1978) is a very interesting account of the rise to the top of the policy agenda of the growth objective in the 1950s and 1960s, and of reaction to claims that continuing economic growth was infeasible due to environmental limits. Useful accounts of debates over limits to growth are to be found in Simon (1981), Simon and Kahn (1984) and Repetto (1985). The October 1998 issue (Vol. 3, part 4) of the journal *Environment and Development Economics* included several papers which revisited the debate over *The Limits to Growth* in response to an article in *The Economist* with the title 'Environmental scares: plenty of gloom'.

McCormick (1989) provides a useful account of the modern history of environmental concerns and their impact on politics, and traces the evolution of the development versus environment debate through the various international conferences which preceded the publication of the Brundtland report. That report (WCED, 1987) is essential reading on sustainable development.

One very important dimension of the sustainability problem, where many particular issues come together and interact, is the matter of feeding the human population. We have not been able to cover this here because of space limitations. The Companion Website discusses some of the issues, and provides pointers to further reading.

Discussion questions

1. Many economists accept that a 'Spaceship Earth' characterisation of the global economy (see Box 1.1) is valid in the final analysis, but would dispute a claim that we are currently close to a point at which it is necessary to manage the economy according to strict principles of physical sustainability. On the contrary, they would argue that urgent problems of malnutrition and poverty dominate our current agenda, and the solution to these is more worthy of being our immediate objective. The objective of physically sustainable management must be attained eventually, but is not an immediate objective that should be pursued to the exclusion of all else.

   To what extent do you regard this as being a valid argument?
2. How effective are measures designed to increase the use of contraception in reducing the rate of population growth?
3. How may the role and status of women affect the rate of population growth? What measures might be taken to change that role and status in directions that reduce the rate of population growth?
4. Does economic growth inevitably lead to environmental degradation?

Problems

1. Use the microeconomic theory of fertility to explain how increasing affluence may be associated with a reduction in the fertility rate.
2. Suppose that families paid substantial dowry at marriage. What effect would this have on desired family size?
3. What effect would one predict for desired family size if family members were to cease undertaking unpaid household labour and undertake instead marketed labour?
4. Take the following data as referring to 2000 (they come from UNDP (2001), P and A are for 1999 and T uses CO_2 data for 1997), and the world as being the sum of these three groups of nations.

	P (millions)	A (PPP US$)	T (tonnes)
Rich OECD	848	26 050	0.0004837
EE and CIS	398	6 290	0.0011924
Developing	4 610	3 530	0.0005382

a. Calculate total world CO_2 emissions in 2000.
b. Work out the 2000 group shares of total population and CO_2 emissions.
c. Assume population growth at 0.5% per year in Rich OECD and EE and CIS and at 1.5% per year in Developing, out to 2050. Assume per capita income growth at 1.5% per year in Rich OECD, at 2.5% per year in EE and CIS, and at 3.0% in Developing, out to 2050. Work out total world emissions and group shares of the total for 2050, and also group shares of world population.

   By what factor does total world emissions increase over the 50-year period?
d. For the same population growth and per capita income growth assumptions, by how much would T have to fall in Rich OECD for that group's 2050 emissions to be the same as in 2000? With Rich OECD emissions at their 2000 level in 2050, assume that T for EE and CIS in 2050 is the same as T for Rich OECD in 2000 (which would be 2050 T for EE and CIS, being about half of its 2000 level) and work out what total world CO_2 emissions would then be.

CHAPTER 3

Ethics, economics and the environment

And God said, Let us make man in our image, after our likeness: and let them have dominion over the fish of the sea, and over the fowl of the air, and over the cattle, and over all the earth, and over every creeping thing that creepeth upon the earth.

Verse 26, Book of Genesis, The Bible, King James Translation

Learning objectives

In this chapter you will

- learn about utilitarianism as the ethical basis for welfare economics
- see how it differs from some other ethical systems
- learn about recent work on measuring utility
- be introduced to some of the criticisms of utilitarianism
- take a first look at the vexed question of discounting
- be introduced to optimal growth analysis where production uses a non-renewable natural resource
- look at the basic model that economists use for thinking about sustainability

Introduction

Environmental and resource economics is concerned with the allocation, distribution and use of environmental resources. To some extent, these matters can be analysed in a framework that does not require the adoption of any particular ethical viewpoint. We can focus our attention on answering questions of the form 'If X happens in a particular set of circumstances, what are the implications for Y?' Analyses of this form constitute what is sometimes described as 'positive' economics.

However, limiting our scope to answering questions of this form is restrictive. Many economists wish also to do 'normative' economics, to address questions about what *should* be done in a particular set of circumstances. To do this it is necessary to use ethical criteria derived from theories about how persons ought to behave. In doing normative economics, generally referred to as 'welfare economics', economists usually employ criteria derived from utilitarian ethical theory. Normative resource and environmental economics is predominantly founded in utilitarian ethics.

The main purpose of this chapter is to provide an introduction to and overview of the nature of the utilitarian approach to ethics, and to show how it informs normative economics. Welfare economics as such is dealt with in Chapters 4 and 11, and gets applied throughout Parts II, III and IV of the book. This chapter begins by looking briefly, in the first two sections, at other approaches to ethics, so as to provide some context. The chapter then, in the third section, sets out the basic elements of utilitarianism as a general approach to the question of how we should behave, and the particular ways in which welfare economics uses that general approach. This section also looks at recent work on measuring utility via surveys of people's self-assessed well-being. Some of the criticisms of utilitarianism and its use in welfare economics are then reviewed in the fourth section of the chapter.

In the context of economic activity and the natural environment, the question of how we should behave with respect to future generations is important. As we saw in the previous chapter, there is, for many, a concern that current economic activity is affecting the environment so as to entail damage to future generations. The fifth section of the chapter looks at the utilitarian approach to the question of intertemporal distribution.

3.1 Naturalist moral philosophies

A fundamental distinction can be drawn between two broad families of ethical systems, humanist and naturalist moral philosophies. In humanist philosophies, rights and duties are accorded exclusively to human beings, either as individuals or as communities – while humans may be willing to give them consideration, non-human things have no rights or responsibilities in themselves. A naturalist ethic denies this primacy or exclusivity to human beings. In this ethical framework, values do not derive exclusively from human beings. Rather, rights can be defined only with respect to some natural system. A classic exposition of this ethic is to be found in Aldo Leopold's 'A Sand County Almanac' (1970, p. 262): 'A thing is right when it tends to preserve the integrity, stability and beauty of the biotic community. It is wrong when it tends otherwise.'

Peter Singer (1993) describes this position as a 'deep ecology' ethic. When a development is proposed, a deep ecologist might argue that the project would not be right if significant disturbances to ecosystems are likely to occur. Given that a large part of human behaviour does have significant ecological implications, strict adherence to a naturalist philosophy would prohibit much current and future human activity. The implications of a thoroughgoing adherence to such a moral philosophy seem to be quite profound, although much depends upon what constitutes a *significant* impact.

A weak form of naturalist ethics – roughly speaking, the notion that behaviour which has potentially large impacts on those parts of the biosphere that are deserving of safeguard, because of their unusualness or scarcity, should be prohibited – has had some impact on public policy in many countries. Examples include the designation of Sites of Special Scientific Interest and the consequent special provisions for management of these sites in the United Kingdom, the system of National Parks in the USA, and the designation of Internationally Important Sites by the World Wide Fund for Nature.

In the period since 1970, a number of important works have emerged which attempt to establish the nature of mankind's obligation to non-human beings. Much of this writing has made use of Kant's categorical imperative, according to which an action is morally just only if it is performed out of a sense of duty and is based upon a valid ethical rule. Justice is not to be assessed in terms of the consequence of an action.

But what is a valid rule? According to Kant, a valid rule is a universal rule. Universality here means that such a rule can be applied consistently to every individual. He writes: 'I ought never to act except in such a way that I can also will that my maxim [rule] should become a universal law.' This principle is Kant's categorical imperative. The basis of ethical behaviour is found in the creation of rules of conduct that each person believes should be universalised. For example, I might legitimately argue that the rule 'No person should steal another's property' is an ethical rule if I believe that everyone should be bound by that rule.

One categorical imperative suggested by Kant is the principle of respect for persons: no person should treat another exclusively as a means to his or her end. It is important to stress the qualifying adverb *exclusively*. In many circumstances we do treat people as means to an end; an employer, for example, regards members of his or her workforce as means of producing goods, to serve the end of achieving profits for the owner of the firm. This is not wrong in itself. What is imperative, and is wrong if it is not followed, is that all persons should be treated with the respect and moral dignity to which any person is entitled.

Kant was a philosopher in the humanist tradition. His categorical imperatives belong only to humans, and respect for persons is similarly restricted. However, naturalists deny that such respect should be accorded only to humans. Richard Watson (1979) begins from this Kantian imperative of respect for persons, but amends it to the principle of respect for others. In discussing who is to count as 'others', Watson makes use of the principle of reciprocity, the capacity to knowingly act with regard to the welfare of others. He denies that only humans have the capacity for reciprocal behaviour, arguing that it is also evident in some other species of higher animal, including chimpanzees, dolphins and dogs. Such animals, Watson argues, should be attributed moral rights and obligations: at a minimum, these should include intrinsic rights to life and to relief from unnecessary suffering.

But many writers believe that human obligations extend to a far broader class of 'others'. The philosopher G.J. Warnock (1971) grappled with the concept of consideration, the circumstances that imply that something has a right for its interests to be taken into account in the conscious choices of others. Warnock concluded that all sentient beings – beings which have the capacity to experience pleasure or pain – deserve to be considered by any moral agent. So for Warnock, when you and I make economic decisions, we have a moral obligation to give some weight to the effects that our actions might have on any sentient being.

Some other naturalist philosophers argue that the condition of sentience is too narrow. Our obligations to others extend beyond the class of other animals that can experience pain and pleasure. Kenneth Goodpaster (1978) concludes that all living beings have rights to be considered by any moral agent. W. Murray Hunt (1980) adopts an even stronger position. He concludes that 'being in existence', rather than being alive, confers a right to be considered by others. For Hunt, all things that exist, living or dead, animate or inanimate, have intrinsic rights.

Although our summary of naturalistic philosophies has been brief, it does demonstrate that the typical humanist philosophy adopted by most economists has not gone unchallenged. It seems to be the case that the moral foundations of some ecological and environmentalist arguments owe much to naturalistic ethics. This may account for why conventional economists and some environmentalists have found it difficult to agree. Readers who wish to explore naturalistic moral philosophy in more depth than has been possible here should consult the Further Reading section at the end of the chapter.

3.2 Libertarian moral philosophy

Libertarianism is a humanist moral philosophy. It takes as its central axiom the fundamental inviolability of individual human rights. There are no rights other than the rights of human individuals, and economic and social behaviour is assessed in terms of whether or not it respects those rights. Actions that infringe individual rights cannot be justified by appealing to some supposed improvement in the level of social well-being. Libertarianism asserts the primacy of processes, procedures and mechanisms for ensuring that fundamental liberties and rights of individual human beings are respected and sustained. Rights are inherent in persons as individuals, and concepts such as community or social rights are not meaningful.

We will discuss the work of one influential libertarian philosopher, Robert Nozick (1974). Nozick's intellectual foundations are in the philosophy of John Locke, and in particular his principle of just acquisition. Locke argued that acquisition is just when that which is acquired has not been previously owned and when an individual mixes his labour power with it. For Locke, this is the source of original and just property rights.

Nozick extends this argument. He asks: when is someone entitled to hold (that is, own) something? His answer is: 'Whoever makes something, having bought or contracted for all other held resources used in the process (transferring some of his holdings for these co-operating factors), is entitled to it.' So any holding is a just holding if it was obtained by a contract between freely consenting individuals, provided that the seller was entitled to dispose of the object of the contract. (Some people will not be entitled to their holdings because they were obtained by theft or deception.) The key point in all of this is free action. Distributions are just if they are entirely the consequence of free choices, but not otherwise.

Libertarians are entirely opposed to concepts of justice based on the consequences or outcomes. An outcome cannot in itself be morally good or bad. Libertarian moral philosophy is likely to drastically limit the scope of what government may legitimately do. For example, policy to redistribute income and wealth (between people, between countries or between generations) in favour of the poor at the expense of the rich requires taxation that is coercive, and so unjust unless every affected person consents to it. Government action would be limited to maintaining the institutions required to support free contract and exchange. Those who believe in a limited role for government have adopted libertarianism enthusiastically. However, it is by no means clear that a completely laissez-faire approach is necessarily implied by libertarianism, as can be seen by considering the following three questions that arise from the notion of just acquisition:

1. What should government do about unjust holdings?
2. How are open access resources to be dealt with?
3. How do external effects and public goods relate to the concept of just acquisition?

If you are unfamiliar with them, the terms open access, external effects and public goods are explained in Chapter 4. You may want to come back to questions 2 and 3 after reading that chapter.

3.3 Utilitarianism

Utilitarianism originated in the writings of David Hume (1711–1776) and Jeremy Bentham (1748–1832), and found its most complete expression in the work of John Stuart Mill (1806–1873), particularly in his *Utilitarianism* (1863). The ethical basis for modern normative economics is a particular variety of utilitarianism, as we shall explain. Utility is the term for the individual's pleasure or happiness introduced by early utilitarian writers. Modern economics still uses this term in that way. The term welfare is used to refer to the social good, which in utilitarianism, and hence welfare economics, is some aggregation of individual utilities. For utilitarians, actions that increase welfare are right and actions that decrease it are wrong.

Utilitarianism is a consequentialist theory of moral philosophy – it is solely the consequences or outcomes of an action that determine its moral worth. In this it differs from motivist theory, according to which an action is to be judged according to its motivation (Kant was a motivist), and from deontological theory, according to which it is an action's inherent nature that makes it right or wrong. For a utilitarian, an action may be considered morally justified even if undertaken for unworthy reasons and having a nature that might in some circumstances be considered bad. For a utilitarian, the ends might justify the means.

3.3.1 Anthropocentric utilitarianism

In order to make a utilitarian judgement we have, among other things, to decide on the composition of the set of entities over whom consequences count. We have to decide who is to be considered in deciding whether an action is right or wrong. The founding fathers of utilitarianism took it as self-evident that only individual humans were morally considerable, that the set of entities over whom consequences should count comprised only human beings. Modern economists adopt the same anthropocentric position. Indeed, in doing applied welfare economics they often restrict the morally considerable set further, and consider only the consequences for the human citizens of a particular nation state.

The restriction to human beings is not a logical necessity. We mentioned earlier a conclusion reached by the philosopher Peter Singer. In his book *Practical Ethics*, Singer adopts what he regards as being a utilitarian position. He argues that utility is derived from gaining pleasure and avoiding pain, and that since all sentient beings (by definition) can experience pleasure or pain, all can be regarded as capable of enjoying utility. Utility, that is, is a characteristic of sentience, not only of humanity. Singer concludes that the principle of judging actions on the basis of their implications for utilities, and hence welfare, is morally valid, but asserts that weight should be given to non-human as well as human utilities.

It needs to be noted here that the rejection of Singer's arguments for the extension of moral considerability need not imply that the interests of non-human entities are ignored. There are two ways in which non-human interests could influence decisions notwithstanding that only human utilities count. First, some humans suffer on account of what they regard as the suffering of non-human, mainly animal, entities. Within the framework of anthropocentric utilitarianism this kind of altruism would entail that what the humans thought the interests of the relevant entities were would be accounted for. Second, humans use some renewable resources – plants and animals – as inputs to production, and prudent resource management would then imply that some consideration be given to, at least, the future availability of such. For both of these sorts of reasons, some species of plants and animals 'have value' to the humans who are directly morally considerable. As we shall see throughout this book, it is often the case that the values arising are not made manifest in markets. An important area of resource and environmental economics is about inducing market systems of economic organisation to take proper account of the ways – direct in the case of altruism, indirect in the case of production use – that what happens to these plants and animals affects human utilities.

3.3.2 Preference satisfaction utilitarianism

Given that we have decided that what is right and wrong is to be decided by the consequences for human individuals, there remains the question of how we should decide which consequences are good, i.e. utility enhancing, and which are bad, i.e. utility diminishing. How should we decide, that is, what is good for people? For the utilitarianism that is the basis of normative economics, the answer is that the affected people decide. If individual A prefers a state of affairs identified by I to a state of affairs identified by II, then according to the preference satisfaction utilitarianism of welfare economics, I confers more utility on A than does II. This is also known as 'the doctrine of consumer sovereignty' – the economy should be ruled by the wants of consumers.

Anthropocentric utilitarianism does not logically entail consumer sovereignty. One could identify individual utility with physical and mental health rather than preference satisfaction. While this is logically true, as a matter of terminological fact most people take utilitarianism to mean self-assessment according to preference. It is precisely because this usage is so widespread that it is important to be clear that an anthropocentric consequentialist theory of ethics does not have to imply consumer sovereignty. What is true is that the preference satisfaction/consumer sovereignty version that economists employ does, as we shall see, lend itself to formalisation and quantification. It is also true, as we shall also see, that it aligns well with the form of economic organisation that has come to dominate human society – the market. It is not, however, without critics, some of whom are economists. We shall look at some of the criticism of (preference satisfaction) utilitarianism after considering how it deals with social welfare.

3.3.3 From utilities to welfare

In utilitarianism, and hence welfare economics, social welfare is some aggregation of individual utilities. For utilitarians, actions that increase welfare are right and actions that decrease it are wrong. We now need to consider precisely how to get from utilities to welfare.

3.3.3.1 Cardinal and ordinal utility functions

One thing that is agreed by all utilitarians is that social well-being, i.e. welfare, is some function of the utilities of all relevant persons. We shall examine shortly what form or forms this function might take. But whatever the answer to this, we can only obtain such an aggregate measure if individual utilities are regarded as comparable across persons.

For an individual, a utility function maps states of the world into a single number for utility. In economics, states of the world are usually represented in terms of levels of the individual's consumption of various goods and services. In that case, we have

$$U = U(X_1, X_2, \ldots, X_i, \ldots, X_N)$$

where U is the utility measure and the arguments of the function X_i are the levels of consumption of the $1, 2 \ldots N$ goods and services. The arguments of a utility function can, however, include, for examples, consumptions by other human economic agents, or states of the environment. We shall be looking at utility functions with these kinds of arguments in the next chapter, and in Chapter 12. For the present, we will work with situations where the arguments are just own levels of consumptions.

We now need to make the distinction between a cardinal and an ordinal measure of utility. Cardinal data are numerical observations where all the standard operations of arithmetic – addition, subtraction, multiplication and division – make sense. Examples of cardinal data are observations on height, weight and length. If John weighs 100 kilos and Jane weighs 70 kilos, it makes sense to say that John is 30 kilos heavier than Jane, and weighs 1.4286 times as much. Ordinal data are numerical observations where ranking is possible, but the standard operations of arithmetic do not apply. Street numbers are an example – from number 10, number 30 is known to be further away than number 20, but it is not known to be twice as far away. Note that we could multiply street numbers by a constant and this would not change their information content – it is only the ordering by number that means anything.

If we want to aggregate meaningfully over individual measures of utility, those measures must be cardinal. Suppose that we have two individuals A and B, that A's utility is 10 and B's is 5, and that it is agreed that simple addition is the way to aggregate. In that case, welfare is 15 utils. But, if the measures are ordinal, we could just as well say that A's utility is 50, in which case welfare is 60 utils. More to the point, it is then 10 times that of B, whereas formerly it was twice that of B. It is as if we were using the highest street numbers used in each case to determine the lengths of two streets.

In doing positive economics, economists have established that preference orderings can be represented by ordinal utility functions, from which the standard propositions of demand theory can be derived. Put this the other way round. Demand theory does not need cardinal measurement of utility, it only needs ordinal measurement. With utility measured ordinally there is no basis for interpersonal utility comparisons. If, as is generally the case, some proposed policy creates winners and losers, we cannot compare the utility gains of winners and losers so as to decide, on utilitarian criteria, that the policy is, or is not, desirable. Equally, across alternative policies, we cannot say which is best.

Given this, most economists would prefer not to have to make interpersonal comparisons when doing normative economics, and have spent some time trying to devise ways of avoiding the need to do so. This area of welfare economics, referred to as compensation tests, is discussed fully in Chapter 4. Here we just sketch the essentials. Suppose that we are considering some change to economic arrangements, some policy, such that the consumption levels for A and B change. If both get to consume more of everything, the standard assumptions of positive demand theory have both better off, and we do not need to make interpersonal comparisons to conclude that welfare is improved by the change. Changes such as this are not typical.

Typically, any proposed change will make one of A or B better off, increase A or B's utility, while making the other worse off, experience lower utility. How now do we decide whether the change is desirable? The obvious thing to do is to add the utility changes, possibly using weights, and see if the answer is positive or negative, concluding that the change is desirable if we get a positive answer. But, if interpersonal utility comparisons are inadmissible, this we cannot do. A way round this is to say that the change is desirable – is welfare improving – if it is such that, according to her evaluation, the gainer from the change would be better off after it and fully compensating the loser, according to his evaluation, for the change. This would be what is known as a 'Pareto improvement' – a change where at least one person gains and nobody loses.

Now, while this way of proceeding does avoid the need for interpersonal comparisons, it is not of much actual use. If economists restricted themselves to advising on policy on the basis of the Pareto improvement test, they would not have a lot to say. They would only be able to say anything about changes where either everybody was a winner, or where the winners had to compensate the losers. Compulsory compensation is not a feature of many policy changes that governments seek advice on. In order to widen the scope for giving advice, economists came up with the idea of the 'potential Pareto improvement' test. According to this, a change is desirable if the gainers could compensate the losers and still be better off. Actual compensation is not required. As shown in Chapter 4, although widely used in applied welfare economics, potential compensation tests do not really solve the problem. If economists want to identify changes that are welfare improving then they need to aggregate over individual utilities, make interpersonal comparisons, and that requires cardinal utility functions. There are basically two ways that economists have responded to this fundamental problem for the practice of normative economics. The first is to opt for a more limited basis on which to offer advice. As discussed in Chapter 4, much of the advice that economists do offer is based on efficiency rather than welfare criteria. The second is to treat utility functions as if they were cardinal and go ahead and work with functions, known as social welfare functions, that aggregate over utilities to produce welfare measures. It is this second approach that we now explore.

3.3.3.2 Social welfare functions and distribution

Consider a hypothetical society consisting of two individuals, A and B, living at some particular point in time. One good (X) exists, the consumption of which is the only source of utility. Let U^A denote the total utility enjoyed by A, and U^B the total utility enjoyed by B, so we have

$$U^A = U^A(X^A)$$
$$U^B = U^B(X^B) \tag{3.1}$$

where X^A and X^B denote the quantities of the good consumed by A and B respectively. We assume diminishing marginal utility, so that

$$U_X^A = \mathrm{d}U^A/\mathrm{d}X^A > 0 \text{ and } U_{XX}^A = \mathrm{d}^2U^A/\mathrm{d}X^{A^2} < 0.$$

and

$$U_X^B = \mathrm{d}U^B/\mathrm{d}X^B > 0 \text{ and } U_{XX}^B = \mathrm{d}^2U^B/\mathrm{d}X^{B^2} < 0.$$

In general, utilitarianism as such does not carry any particular implication for the way output should be distributed between individuals in a society. Generally, social welfare, W, is determined by a function of the form

$$W = W(U^A, U^B) \tag{3.2}$$

where $W_A = \partial W/\partial U^A > 0$ and $W_B = \partial W/\partial U^B > 0$ so that social welfare is increasing in both of the individual utility arguments. Here, welfare depends in some particular (but unspecified) way on the levels of utility enjoyed by each person in the relevant community. Given cardinal utility measurement, such a social welfare function (SWF) allows us to rank different configurations of individual utilities in terms of their social worth.

Assume, to make things simple and concentrate on the essentials here, that there is a fixed total quantity of the good, denoted $\bar{X}$. (Analysis of cases where the total quantities of two consumption goods are variable are considered in Chapter 4.) Consumption, and hence utility levels, for A and B are chosen so as to maximise welfare. X^A and X^B are chosen, that is, to maximise

$$W = W(U^A, U^B)$$

given U^A and U^B determined according to equations (3.1) and subject to the constraint that $X^A + X^B = \bar{X}$.

It is shown in Appendix 3.2 (using the Lagrange method outlined in Appendix 3.1) that the solution to this problem requires that

$$W_A U_X^A = W_B U_X^B \tag{3.3}$$

This is the condition that the marginal contributions to social welfare from each individual's consumption are equal. What this means is that the consumption levels for each individual will vary with the utility function for each individual and with the nature of the social welfare function (3.2).

A widely used particular for the function (3.2) has W as a weighted sum of the individual utilities as in

$$W = w_A U^A(X^A) + w_B U^B(X^B) \tag{3.4}$$

where w_A and w_B are the, fixed, weights. These weights reflect society's judgement of the relative worth of each person's utility. In this case the condition for the maximisation of social welfare is

$$w_A U_X^A = w_B U_X^B$$

A further specialisation is to make the weights equal to one, so that social welfare is a simple sum of utilities of all individuals. For this special case we have

$$W = U^A + U^B \tag{3.5}$$

Figure 3.1 illustrates one indifference curve, drawn in utility space, for such a welfare function. The social welfare indifference curve is a locus of combinations

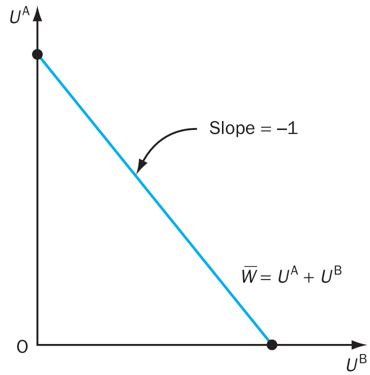


Figure 3.1 An indifference curve from a linear form of social welfare function

of individual utilities that yield a constant amount of social welfare, $\bar{W}$. The assumption that the welfare function is additive in utilities implies that the indifference curve, when drawn in utility space, is linear.

In the case of equal welfare weights the condition for the maximisation of social welfare, equation 3.3, becomes

$$U_X^A = U_X^B \tag{3.6}$$

which is the equality of the individuals' marginal utilities. This still does not tell us how goods should be distributed. To find this, we need some information about the utility function of each individual. Consider the case where each person has the same utility function. That is

$$U^A = U^A(X^A) = U^B(X^B)$$
$$U^B = U^B(X^B) = U^A(X^A) \tag{3.7}$$

It is then easy to see that in order for marginal utility to be equal for each person, the consumption level must be equal for each person. An additive welfare function, with equal weights on each person's utility, and identical utility functions for each person, implies that, at a social welfare maximum, individuals have equal consumption levels.

The solution to the problem with equal weights and identical utility functions is illustrated in Figure 3.2. Notice carefully that the diagram is now drawn in commodity space, not utility space. Under the common assumption of diminishing marginal utility, the linear indifference curves in utility space in Figure 3.1 map into welfare indifference curves that are convex

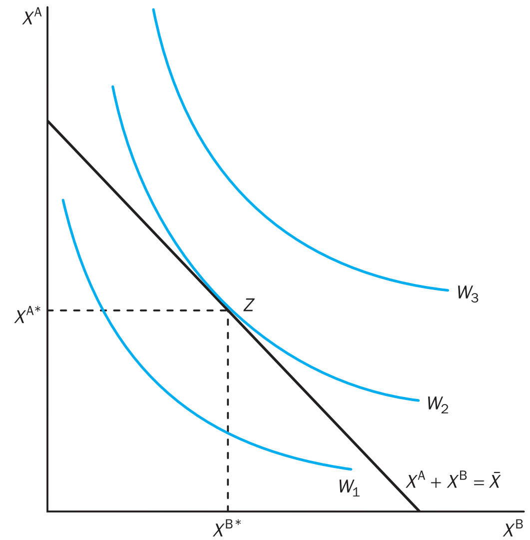


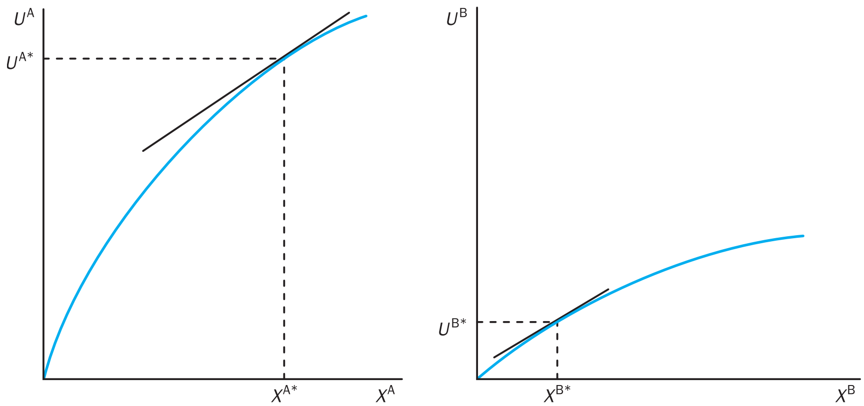
Figure 3.2 Maximisation of social welfare subject to a constraint on the total quantity of goods available

from below in commodity space. The curves labelled W_1, W_2 and W_3 are social welfare indifference curves, with $W_1 < W_2 < W_3$. Remember that we assume that there is a fixed quantity of the good $\bar{X}$ available to be distributed between the two individuals. Maximum social welfare, W_2, is attained at the point Z where the consumption levels enjoyed by each person are X^{A*} and X^{B*}. The maximised level of social welfare will, of course, depend on the magnitude of $\bar{X}$. But irrespective of the level of maximised welfare, the two consumption levels will be equal.

In the example we have just looked at, the result that consumption levels will be the same for both individuals was a consequence of the particular assumptions that were made. Utilitarianism does not generally imply an equal distribution of goods. An unequal distribution at a welfare maximum may occur under any of the following conditions:

1. The SWF is not of the additive form specified in equation 3.4.
2. The weights attached to individual utilities are not equal.
3. Utility functions differ between individuals.

To illustrate the third condition, suppose that the utility functions of two persons, A and B, are as shown in Figure 3.3. The individuals have different



At X^{A*} and X^{B*}, $\frac{dU^A}{dX^A} = \frac{dU^B}{dX^B}$

Figure 3.3 Maximisation of social welfare for two individuals with different utility functions

utility functions in that A enjoys a higher level of utility than individual B for any given level of consumption. We still assume that the social welfare function is additive with equal weights, so that equal marginal utilities are required for welfare maximisation. Because of the difference in the utility functions, the marginal utilities of the two individuals can only be equal at different levels of consumption. In interpreting the diagram, recall that the value of marginal utility at a particular level of consumption is indicated by the slope of the (total) utility function at that point.

The outcome shown in Figure 3.3 illustrates something of a paradox. Individual B is less efficient at turning consumption into utility than individual A. The result then of using a simple addition of utilities is that she gets less allocated to her. Suppose that B is that way because she suffers from depression. Would we then want to say that utilitarianism with equal weights is fair? This illustrates an important general point about ethical theorising. As well as considering the apparent desirability of the adopted principles as such, it is also necessary to consider explicitly what the principles imply in different circumstances.

3.3.4 Measuring (cardinal) utility

As noted, the standard position in modern economics is that utility cannot be measured cardinally. However, since the 1950s social scientists have been using surveys to ask people about their own assessments of their well-being, or utility, and in the past decade or so some economists have come to the view that this work provides cardinal data with which the sources of well-being can be investigated. We now briefly report on work in this area: readers who wish to learn more about developments in this area will find references in the Further Reading section at the end of the chapter.

A typical question about a survey respondent's subjective well-being, SWB, would be

> All things considered, how satisfied are you with your life as a whole these days?

with the respondent being required to indicate their answer on a scale ranging from 1 (or 0) for very dissatisfied to 10 for very satisfied. Hundreds of surveys asking this type of question have now been conducted in some 100 countries, and the results subjected to a lot of analysis. The great majority of the social scientists, and now economists, who have looked at this work seriously consider that this type of question does generate meaningful information which can be treated as cardinal data on SWB – they consider, for example, that it makes sense to average the scores across respondents to get an estimated score for the population from which respondents are drawn. Many of the surveys ask respondents about their socioeconomic characteristics, and regression analysis is used to establish statistical relationships between SWB and such characteristics.

Most of the works in this growing field do not use ‘SWB’, ‘satisfaction’, or ‘utility’ in their title. Most do use the word ‘happiness’, and the field is generally referred to as ‘happiness research’. In many of the surveys, rather than that given above, the question put to respondents is

> All things considered, how happy would you say you are with your life these days?

with respondents being required to give an answer in the range 1 (or 0), for very unhappy, to 10 for very happy. One of the reasons that people working in the field judge that these questions generate useful, cardinal, information is that when used in the same context they give rise to very similar scores.

There are other reasons for thinking that happiness/satisfaction/utility can be measured cardinally. One way in which such surveys have been evaluated is by asking the same people the same questions at different points in time. If an individual’s circumstances do not change, then her happiness score should not change significantly – this is what has been found to be the case. Another test of these subjective well-being measures is to see if they correlate with other indicators of happiness. It has been found that, compared with the average person, individuals with higher than average self-assessed happiness scores are, for example, more likely to:

- be rated as happy individuals by family and friends;
- be more optimistic about the future;
- be less likely to attempt suicide;
- recall more positive than negative life events;
- smile more during social interaction;
- be more healthy.

As noted above, these surveys have been done in many countries. An obvious concern would be that ‘happy’ might mean different things in different languages. This appears not to be a major problem. Country average scores have, for example, been based on three different approaches – asking people how happy they are, asking people how satisfied they are, and asking them to give their lives a score on a scale running from ‘worst possible life’ to ‘best’. It was found that the rankings of countries were almost identical across the three different bases for the national average scores.

Finally there is the fact that variations in individuals’ scores are not random noise but are correlated with various individual attributes. Individuals differ in their genetic predispositions in relation to feeling happy or otherwise. This is a matter of everyday observation, which is confirmed in experiments and surveys. By use of proper statistical methods, the correlation between the happiness score and attribute x score is calculated after allowing for, or controlling for, the influence of other attributes. Only those attributes that are recorded in the survey can be controlled for. Genetic make-up is not recorded in the survey, and is not controlled for. This is not considered to be a problem for using such surveys to study how life circumstances affect happiness, as genetic make-up is thought to affect the overall disposition to happiness, rather than the responses to particular life circumstances. It is thought, for example, to mean that A will be happier than B for exactly the same life circumstances, rather than to mean that more of x makes A happier but B less happy.

Across individuals, the following have been found to be positively associated with SWB: absolute income, income relative to that of others, current income relative to past income, being married, membership of a group (such as a religious group), political participation, good physical health. Job insecurity and unemployment have been found to be negatively related to SWB. Statistical association is not the same as causation and in some cases the possibility that causation runs from happiness to the attribute cannot be entirely ruled out. Thus, for example, it is possible that being genetically predisposed to happiness may make one more attractive and, hence, more likely to be married and employed. However, in many cases, there is evidence, for example from surveys that track the same respondents over time, that there is causation from the life circumstance to the level of SWB.

3.3.4.1 SWB and income

We now look at the relationship between income and happiness. This can be done in three ways. We can look at the happiness–income relationship across

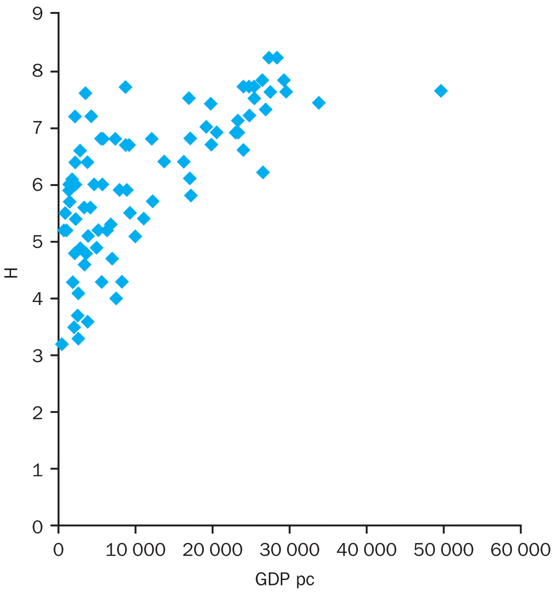


Figure 3.4 Happiness and income across countries

individuals in a given country, across country averages, or over time in a given country.

Looking first at cross-country data, Figure 3.4 shows a plot of national average happiness scores against GDP per capita. The happiness data, vertical axis, are based on surveys conducted in the period 1995–2005 and were taken from the World Data Base of Happiness (see Further Reading for reference) where there are data for 90 countries, across which the average score was 6.17 with a minimum of 3.2 and a maximum of 8.2. The GDP data are for per capita GDP in 2000 measured in PPP $s, and were taken from the *Human Development Report 2002*. From these data, increasing income increases SWB at a decreasing rate – at high levels of GDP per capita a given increase in it produces a smaller increase in happiness than it does at low levels. These data are consistent with a declining marginal utility of income.

If we look at data on individuals in a given country at a given point in time, we find that individual happiness increases with individual income. However, most such studies for advanced economies find that as income increases so the happiness index increase associated with a given increase in income decreases. The relationship is essentially the same form as that shown in Figure 3.4 – there are decreasing returns to income increases. The evidence for declining marginal utility is less strong in studies of developing economies.

Finally we consider what happens to the national average happiness score as national income grows over time. Figure 3.5 shows the trends in GDP per capita and the percentage of people reporting

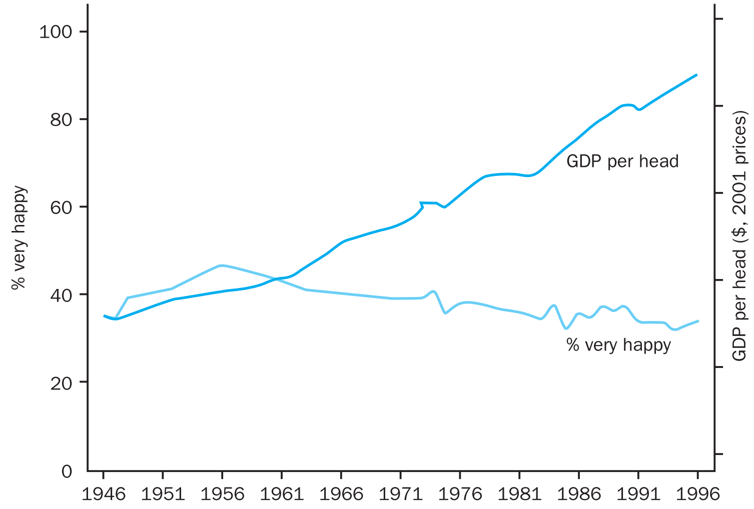


Figure 3.5 Post-World War II trends in happiness and GDP per capita in the USA
Source: Layard (2003)

themselves as 'Very Happy' in the USA for the period 1946 to 1996. GDP per capita increased steadily, while the very happy proportion actually fell slightly. Essentially there is no relationship between average income as measured by GDP and this measure of national happiness. This is what turns out to be the case for virtually all of the rich economies for which data are available over a few decades or more – GDP per capita grows steadily while national average happiness remains effectively unchanged.

There is a paradox here. If we look across individuals, or across countries, at a point in time, then increasing income goes with increasing happiness, albeit at a declining rate as income increases. On the other hand, when we look at a whole rich economy over time, rising per capita GDP does not go with increasing happiness. Surely, if having more income makes an individual happier, individuals on average becoming richer over time should mean individuals on average becoming happier over time.

Table 3.1 re-states this paradox. It refers to the USA, but what it reports is what is found in the recent history of many rich economies. A higher percentage of rich people than of poor people report themselves as 'Very happy', and a higher percentage of poor people than of rich people report themselves as 'Not too happy', in both 1975 and 1998. At a point in time, those with higher incomes are happier. However, despite the incomes of both rich and poor growing at something like 2% per year for over 20 years, the proportions for 'Very happy', 'Pretty happy' and 'Not too happy' were much the same, for both groups, in 1998 as they were in 1975. Average income grew a lot, but that did not much affect the proportions of responses, either at the top or at the bottom of the income distribution.

Table 3.1 Percentages reporting various states of happiness by income group, USA

	Proportion of people with incomes in the top quarter of the range %		Proportion of people with incomes in the bottom quarter of the range %	
State reported	1975	1998	1975	1998
Very happy	39	37	19	16
Pretty happy	53	57	51	53
Not too happy	8	6	30	31

Source: Layard (2003)

3.3.4.2 Adaptation, aspirations, and interdependencies

We now sketch the outlines of a resolution of the paradox that appears to be widely agreed on in the happiness literature. What follows here is necessarily somewhat superficial, though we think it conveys the essentials; the Further Reading section provides guidance to materials which deal more fully with the issues raised here, and provide extensive references to the original sources.

The first basic idea is that an individual's utility depends on the relationship between her aspirations and the corresponding outcomes for the various aspects of her life, and on how her outcomes compare with outcomes for others. The second basic idea is that her aspirations depend on her own past experiences in the relevant areas. The process by which aspirations are related to own experience is known as 'adaptation', or 'habituation'.

Consider one of n individuals, arbitrarily labelled 1, for whom the utility function for period t could be written

$$U_{1t} = U_1(f_1\{A_{11t}, E_{11t}\}, f_2\{A_{12t}, E_{12t}\}, \ldots f_j\{A_{1jt}, E_{1jt}\} \ldots f_m\{A_{1mt}, E_{1mt}\}:$$
$$g_1\{E_{11t}, E_{O1t}\}, g_2\{E_{12t}, E_{O2t}\} \ldots g_j\{E_{1jt}, E_{Ojt}\} \ldots g_m\{E_{1mt}, E_{Omt}\})$$

where there are m relevant areas (income, health, marital status, etc.), where A stands for aspiration and E for experience, where f_j is the function that maps the relationship between A_{1j} and E_{1j} into an argument in the utility function, where the subscript O is for 'others', and g_j is the function that maps the relationship between E_{1j} and E_{Oj} into an argument in the utility function. All individuals are assumed to have utility functions of this form. This represents the first idea. For this individual and looking at the jth relevant area, the second idea can be represented by the function

$$A_{1jt} = h_{1j}(E_{1jt-1}, E_{1jt-2}, \ldots E_{1jt-T})$$

where $t - 1$ is the previous period, $t - 2$ the one before that, and so on. This says that the current jth aspiration level for this individual depends on her past experience in regard to j.

One finding from the empirical studies is that the completeness and speed of adaptation varies across j. For health, for example, it appears that a major deterioration has a permanent effect on $U - A$ for health remains significantly different from E. For income, or material consumption, on the other hand, it appears that adaptation is relatively quick and complete – the current level of an individual's A quickly and fully comes to equal the recent past levels of that individual's E. Also, for income it is the case that the E's for other individuals have an influence on U for individual 1. This is an example of 'interdependence', also known in this literature as 'rivalry' in connection with income and material consumption.

The paradox is now resolved as follows. A rise in individual 1's income permits a new higher level of consumption, in the form of new clothes, say. With E_{11} for 1's consumption, initially this gives a higher level of E_{11}, and hence, for given A_{11} and E_{O1}, a higher level of utility. But she soon gets used to the new clothes, and A_{11} adjusts to past E_{11}, moving U_1 back toward to its former level. In a widely used phrase, 'the novelty wears off'. Also, in a growing economy, the consumption of others will also be rising, restoring the gap between E_{11} and E_{O1} to its former level, and again moving U_1 back toward its former level.

Together, adaptation and rivalry explain why at a point in time individuals with higher incomes are usually happier than those with lower incomes, although over time generally rising incomes do not produce generally increasing happiness. Both adaptation and rivalry – 'keeping up with the Joneses' – are consistent with everyday experience, though their implications for what economic growth can deliver in terms of happiness are perhaps not widely appreciated. Mainstream economics largely ignores the results from the happiness literature, working with utility functions which are, in the above notation:

$$U_{1t} = U_1(E_{11t}, E_{12t}, \ldots E_{1jt}, \ldots E_{1mt})$$

3.3.4.3 Implications of happiness research results

Clearly, happiness research results have implications for positive economics, and for utilitarian normative, or welfare, economics. In neither case have the implications been much explored as yet, precisely because mainstream economics has not really taken the results 'on board' yet, as perusal of a standard economics text or journal will confirm. Clearly, if this work has implications for welfare economics, it will have implications for the policy advice offered by environmental and resource economics, which is based on mainstream welfare economics. In neither case has much work yet been done on the implications of happiness research results for standard results in the literature, and these are important areas for investigation. Having said that, we can say a few things here.

First, the results clearly have implications for thinking about the benefits of economic growth in advanced economies. This is not a question that we have a lot to say about in this book, though it is clearly an important one in relation to the issues addressed in the previous chapter, and with respect to sustainable development. We will, in Chapter 19, look at a suggestion for using the results of SWB surveys, with other information, as a way of measuring national economic performance in relation to the objective of sustainable development.

Second, the results imply that interpersonal interdependencies are pervasive, whereas welfare economics assumes that they are rare. As we will consider in Chapter 4, the market system fails when there are interpersonal dependencies, otherwise known as 'externalities'. Welfare economics is largely about finding ways to correct for market failure, but it does this on the basis that its sources are rare. The ideal world that provides the goal that policy aims at is one where there are no interpersonal interdependencies. Given that they are, in fact, ubiquitous, one might wonder if a different approach to economic policy analysis might not be more appropriate.

At the particular level, we provide, by way of illustration, one example of a widely accepted policy prescription that has been disputed on the basis of the results from happiness research. It is taken from a paper, Layard (2005b), that considers some revisionist policy prescriptions arising from results in happiness research: see also Layard (2005a). Income tax is, according to standard public sector economics, a regrettable necessity. It is bad because it 'distorts' the choice between consumption and leisure in favour of the latter, by taxing, in effect, the former but not the latter. With income taxation, it follows, people work less than is required for welfare maximisation. Ideally, public sector expenditure should be financed by lump-sum taxation which does not distort the work/leisure choice. However, lump-sum taxation is infeasible, so income tax is needed: the point is to keep it as low as possible.

This conclusion depends on there being no externalities involved in the work (to consume) leisure choice. But from happiness research results we know that in fact there are externalities involved. The amount that I work and consume influences the benefit that others get from their work and consumption, and vice versa. I work more to keep up with the Joneses and they work more to keep up with me, and we all work more than would maximise total utility. In that case, some income taxation is desirable as, far from being a distortion, it acts to correct the problem caused by the externality. It is like the taxes on environmental pollution that we will be discussing later in this book: see Chapters 4 and 5 especially.

There is research work yet to be done, and papers to be written, about whether, and how, the policy prescriptions of environmental and resource economics would be affected if analysis assumed utility functions that incorporate interpersonal rivalries, aspirational effects, adaptation and habituation.

3.4 Criticisms of utilitarianism

There is, of course, much criticism of the utilitarian approach to ethical theory. As noted already, there are other, non-consequentialist, theories of ethics, which criticise utilitarianism if only by implication. In this section we look first at one influential recent contribution to moral philosophy which is concerned with utilitarianism generally, and then at some criticisms directed primarily at the preference satisfaction utilitarianism that is the basis for modern welfare economics.

3.4.1 Rawls: a theory of justice

The work of John Rawls in *A Theory of Justice* (1971) has influenced the consideration given by economists to ethical issues. Rawls's work challenges classical utilitarianism, where welfare is the simple sum of individual utilities. His objection is grounded in the following assertion. Being indifferent to the distribution of satisfaction between individuals (and only being concerned with the sum of utilities), a distribution of resources produced by maximising welfare could violate fundamental freedoms and rights that are inherently worthy of protection.

In common with many moral philosophers, Rawls seeks to establish the principles of a just society. Rawls adopts an approach that owes much to the ideas of Kant. Valid principles of justice are those which would be agreed by everyone if we could freely, rationally and impartially consider just arrangements. In order to ascertain the nature of these principles of justice, Rawls employs the device of imagining a hypothetical state of affairs (the 'original position') prior to any agreement about principles of justice, the organisation of social institutions, and the distribution of material rewards and endowments. In this original position, individuals exist behind a 'veil of ignorance'; each person has no knowledge of his or her inherited characteristics (such as intelligence, race and gender), nor of the position he or she would take in any agreed social structure. Additionally, individuals are assumed to be free of any attitudes that they would have acquired through having lived in particular sets of circumstances. The veil of ignorance device would, according to Rawls, guarantee impartiality and fairness in the discussions leading to the establishment of a social contract. Rawls then seeks to establish the nature of the social contract that would be created by freely consenting individuals in such an original position.

He reasons to the conclusion that, under these circumstances, people would unanimously agree on two fundamental principles of justice. These are

> First: each person is to have an equal right to the most extensive basic liberty compatible with a similar liberty for others.
>
> Second: social and economic inequalities are to be arranged so that they are both (a) reasonably expected to be to everyone's advantage, and (b) attached to positions and offices and open to all. . . . [The Difference Principle]

It is the second principle that is of interest here. The Difference Principle asserts that inequalities are only justified if they enhance the position of everyone in society (if they lead to Pareto improvements).[1] The Difference Principle has been interpreted as a presumption in favour of equality of position; deviations from an equal position are unjust except in the special cases where all persons would benefit (or perhaps where the least advantaged benefit). Economists have used the utilitarian framework to present the Rawlsian position in two ways.[2] One involves giving the SWF a particular form, the other involves giving the utility function a particular form. We now look briefly at each approach.

3.4.1.1 A Rawlsian Social Welfare Function

One approach has been to argue that a Rawlsian position can, for the case of two individuals, be represented by a SWF of the form:

$$W = \min(U^A, U^B) \tag{3.8}$$

This says that W is equal to whichever is the smaller of U^A and U^B, that W is the minimum of U^A and U^B. Two SWF indifference curves from such a function are illustrated in Figure 3.6. As the utility level of the least advantaged person determines welfare, a Rawlsian SWF implies that raising the utility of the person with the lowest utility level will increase welfare. Compare the two points labelled b and c in Figure 3.6, which by virtue of lying on one indifference curve generate identical levels of social welfare. Starting from point b, reallocate utility between persons, by subtracting $\{b - d\}$ utility from person A and adding this to person B. The point labelled e will have been attained on another indifference curve with a higher level of social welfare. It is clear that the only combinations of utility for which higher welfare levels are not possible through interpersonal transfers of utility are those which lie along the 45° ray from the origin. Along this locus, utility is allocated equally between individuals. So for any given total amount of utility, a Rawlsian social welfare

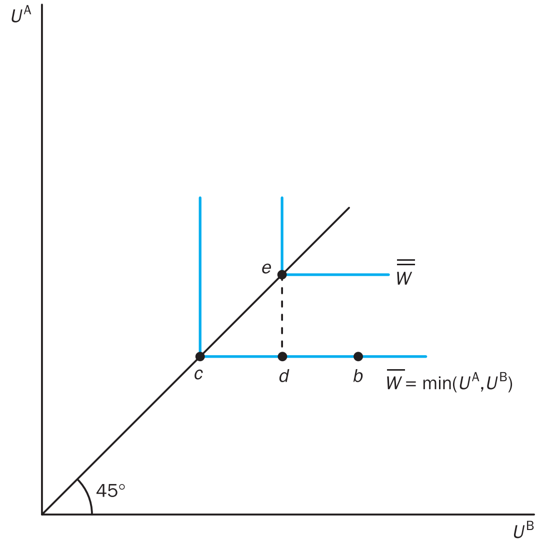


Figure 3.6 Rawlsian social welfare function indifference curves

function implies that, whenever utility levels differ between individuals, it is possible to increase social welfare by redistributing utility from individuals with higher utility to those with lower utility. An egalitarian distribution is implied by this logic.

3.4.1.2 Iso-elastic utility functions

The second approach to representing a Rawlsian concern for the worst-off within a utilitarian framework retains the simple additive welfare function, but specifies the utility functions in a particular way. Assume that all individuals have the same utility function:

$$U = \frac{X^{1-\eta}}{1-\eta} \quad \text{for } \eta > 0 \quad \text{and } \eta \neq 1 \tag{3.9}$$

This form for the utility function is widely used in welfare economics, and is known as an iso-elastic utility function because the elasticity of marginal utility with respect to consumption X is a constant, equal to the parameter η.

[1] Rawls sometimes seems to advocate a rather different position, however, arguing that inequalities are justified in particular when they maximally enhance the position of the least advantaged person in society.

[2] It seems likely that Rawls himself would disapprove of forcing his theory into a strict utilitarian framework.

For equation 3.9

$$U_X = \frac{dU}{dX} = X^{-\eta} \text{ and } U_{XX} = \frac{d^2U}{dX^2} = -\eta X^{-\eta-1}$$

and substituting in the definition for the elasticity of marginal utility as

$$-U_{XX} \cdot X \big/ U_X$$

gives

$$-\frac{-\eta X^{-\eta-1} \cdot X}{X^{-\eta}} = \eta$$

For $\eta = 1$, equation 3.9 would give U as infinity for all X, which is why the restriction $\eta \neq 1$ appears. The iso-elastic utility function for the, constant, elasticity of marginal utility equal to 1 is

$$U = \ln X \tag{3.10}$$

with $U_X = X^{-1}$ and $U_{XX} = -X^{-2}$.

With equation 3.9 for the utility function, the two-person SWF is

$$W = \frac{(X^A)^{1-\eta}}{1-\eta} + \frac{(X^B)^{1-\eta}}{1-\eta} \tag{3.11}$$

and for equation 3.10 it is

$$W = \ln X^A + \ln X^B \tag{3.12}$$

Table 3.2 shows the weights that these SWFs apply to increases in consumption for each of two individuals, where A is 10 and 100 times better off than B in consumption terms. The weights are the partial derivatives with respect to X_A and X_B. The entries in the row for $\eta = 1$ are actually calculated using equation 3.12. In Table 3.2, the relative weight accorded to consumption increases for the worst-off individual increases as the degree of inequality between the individuals increases, and as η gets bigger. This is demonstrated generally in Appendix 3.2. It is generally true, as illustrated in Table 3.2, that:

for $\eta = 0$, the SWF treats an extra unit of consumption equally across individuals – in Table 3.2, an increase of 1 in consumption contributes 1 to ΔW irrespective of who gets it.

for $\eta = 1$, the SWF treats equal proportional increases in consumption equally across individuals – in Table 3.2, for $X^A = 10$ and $X^B = 1$, 1% increases are 0.1 and 0.01 which both contribute 0.01 to ΔW.

for $\eta > 1$, the SWF treats an x% increase in consumption for a poorer person as more valuable than an x% increase in consumption for a better off person – in Table 3.2, for $X_A = 100$ and $X_B = 1$, 1% increases are 1 and 0.01 which for $\eta = 2$ contribute 0.0001 and 0.01 respectively to ΔW.

With iso-elastic utility functions, so that the SWF takes the form of equation 3.11, the Rawlsian position arises as η goes to infinity – small improvements for the worst-off carry much more weight than large improvements for the best-off.

Table 3.2 Welfare weights for consumption increases

	$X^A = 10, X^B = 1$		$X^A = 100, X^B = 1$	
η	W_{XA}	W_{XB}	W_{XA}	W_{XB}
0	1	1	1	1
0.25	0.5623	1	0.3162	1
0.50	0.3162	1	0.1000	1
0.75	0.1778	1	0.0316	1
1.0	0.1000	1	0.0100	1
1.5	0.0316	1	0.0010	1
2.0	0.0100	1	0.0001	1
3.0	0.0010	1	0.000001	1
4.0	0.0001	1	0.00000001	1

3.4.2 Criticisms of preference based utilitarianism

The basic idea involved in the version of utilitarianism that is used in welfare economics is that individuals' preferences are the measure of social welfare. Subject to the constraints given by the availability of resources and technological possibilities, people should get what they want. Social welfare improves when people get more of what they want. Economics does not inquire into the determinants of individuals' preferences, which are taken as given. Economics does not ask questions about what is good for people. The answer to such questions implicit in economics is that individuals are the best judge of what is good for themselves, so that their preferences tell us what is good for them.

Criticism of this, consumer sovereignty, approach to social welfare has come from some economists, as well as many non-economists. Not all of the criticism is well founded. One frequently finds non-economists claiming that economics assumes that individuals only care about their own consumption of produced goods and services. This claim is wrong, though some introductory economics texts do not do very much to counter the impression that supports it. In fact, the utility functions used in welfare economic analysis can, and do, include, for example, arguments which are indicators of the state of the environment. What is true is that market systems do not work very well where individuals have preferences over things other than their own consumption of produced goods and services. But, as we shall be working through in many subsequent chapters, one of the major concerns of welfare economics is to devise policies to make market systems work better in such circumstances. In regard to preferences over the state of the environment, economists have devised a whole range of policies and techniques which they claim can make market systems perform better according to consumer sovereignty criteria.

Critics of consumer sovereignty are on firmer ground when, on informational grounds, they question the assumption that people always know what is good for them, that their preferences reflect their true interests. The questions raised can be broadly classified as being of two kinds. First, taking preferences as given and truly reflecting interests, is it reasonable to assume that people generally have enough information to properly assess the consequences for their own utility of the various alternatives open to them? Second, is it reasonable to assume generally that, in a world where socialisation processes and advertising are pervasive, peoples' preferences do truly reflect their interests? The 'happiness' research introduced in Section 3.3.4 above, 'Measuring (cardinal) utility', suggests negative short answers to these questions: readers interested in longer answers should follow some of the relevant suggestions in the Further Reading section at the end of the chapter. Some aspects of these questions as they arise in the particular context of applying welfare economics to environmental issues will be raised in Chapters 11 and 12.

One economist who has written extensively about the utilitarian basis for economics is the Nobel laureate Amartya Sen – see especially Sen (1987). According to Sen, persons have a fundamental dualism, being concerned with the satisfaction of their own preferences and also pursuing objectives which are not exclusively self-interested. Individuals exist, that is, as both 'consumers' and 'citizens'. In regard to concern for others, altruism, for example, Sen distinguishes between 'sympathy' and 'commitment'. Sympathy is where my concern is reflected in the arguments of my utility function, so that if some change improves the lot of the relevant other(s) my own utility increases. Commitment is where my concern is based on my ethical principles, and to the extent that I am committed to other(s) I may approve of some change even though it reduces my own utility. For some people, that is, activity may be directed to pursuing goals that do not affect the arguments of their utility functions. This does not in itself imply that utilitarianism should be abandoned, but rather that its practice is more problematic than many economists recognise. We shall return to this sort of argument in relation to social decisions about the environment in Chapter 11.

3.5 Intertemporal distribution

Many of the issues with which we deal in this text involve choices with consequences that extend over time. Such choices are said to have an 'intertemporal' dimension. Where we deal only with current consequences we are doing an 'intratemporal' analysis. Thus far in this chapter we have been looking at utilitarianism as the ethical basis for intratemporal normative economics. Most economists also approach normative intertemporal issues on the basis of utilitarianism, and here we want to provide an introductory overview of the general approach adopted by most economists, and introduce some of the main issues. We shall revisit intertemporal welfare economics in Chapter 11, where we will see that while there is widespread agreement among economists about the general, utilitarian, approach to be followed, its implementation can give rise to disagreement and disputation among economists. Many non-economists take strong exception, especially in regard to matters environmental, to the economic approach to intertemporal problems. Particularly, many who are concerned for the environment take strong exception to the 'discounting' of future consequences, which we discuss here and in Chapter 11.

In order to keep the analysis simple, and to focus clearly on intertemporal ethics, we will assume that the size of the human population is constant over time, and that we can think about the interests of everybody in each year in terms of the utility function of the 'representative individual' for each year. We will assume that the utility function is invariant over time, though of course the levels taken by its arguments may vary over time. These are very strong simplifying assumptions, but they do make it possible to bring out some important issues reasonably clearly.

3.5.1 The utilitarian intertemporal social welfare function

In the intertemporal case as in the intratemporal case distributional issues are examined by looking at the maximisation, subject to the appropriate constraints, of the function that maps utilities into welfare. We begin, therefore, with the specification of the intertemporal social welfare function. Note that given that we are going to work with a social welfare function that aggregates utilities at different dates, it follows that we are assuming that utilities are cardinally measurable.

Initially, we consider just two years so that we can use the same general form of notation as when looking at two individuals at a point in time. Year 0 is the current year; year 1 is next year. Then U_0 and U_1 denote the utility enjoyed by the representative individual in years 0 and 1, respectively. W now denotes *intertemporal* social welfare. In general terms the intertemporal social welfare function can then be written as:

$$W = W(U_0, U_1)$$

The specific functional form usually employed by utilitarianism is

$$W = \varphi_0 U_0 + \varphi_1 U_1 \tag{3.13}$$

so that W is a weighted average of the utilities for each year, where ϕ_0 and ϕ_1 are the weights used in summing utility over years to obtain a measure of social welfare. The utilitarian approach to intertemporal questions is typically further specialised by having the weights in equation 3.13 take a particular form. It is usual to set $\phi_0 = 1$ and $\phi_1 = 1/(1 + \rho)$, where ρ is the utility discount rate. Equation 3.13 then becomes

$$W = U_0 + \frac{U_1}{1 + \rho} \tag{3.14}$$

Time discounting, for $\rho > 0$ as generally assumed, means that future utility counts for less than, is 'discounted' with respect to, the same quantity of present utility in obtaining a measure of total welfare over time. In this formulation, the value of a small increment of utility falls, is discounted, as its date of receipt is delayed. For $\rho = 0$ current and future utility would be equally weighted.

Before looking at the justification for this kind of discounting, it will be useful to note some generalisations and modifications of the foregoing that are widely encountered in the literature, and will be used subsequently in this book. For T years, the intertemporal social welfare function is

$$\begin{aligned} W &= \frac{1}{(1+\rho)^0}U_0 + \frac{1}{(1+\rho)^1}U_1 + \ldots + \frac{1}{(1+\rho)^T}U_T \\ &= \sum_{t=0}^{t=T} \frac{1}{(1+\rho)^t}U_t = \sum_{t=0}^{t=T} \varphi_t U_t \end{aligned} \tag{3.15}$$

where $\varphi_t = (1 + \rho)^{-t}$. In many of the problems that we shall be investigating in later chapters, an infinite time horizon will be used, in which case equation 3.15 will be

$$W = \sum_{t=0}^{t=\infty} \frac{1}{(1+\rho)^t}U_t = \sum_{t=0}^{t=\infty} \varphi_t U_t \tag{3.16}$$

It will often be convenient to work with the continuous time version of equation 3.16, which is

$$W = \int_{t=0}^{t=\infty} U_t e^{-\rho t} dt = \int_{t=0}^{t=\infty} U_t \varphi_t dt \tag{3.17}$$

where the weights are $\varphi_t = e^{-\rho t}$.

In this specification of the intertemporal social welfare function, the weights attached to each year's utility, the φ's, decline with time at a constant exponential rate, as illustrated in Figure 3.7. The constant

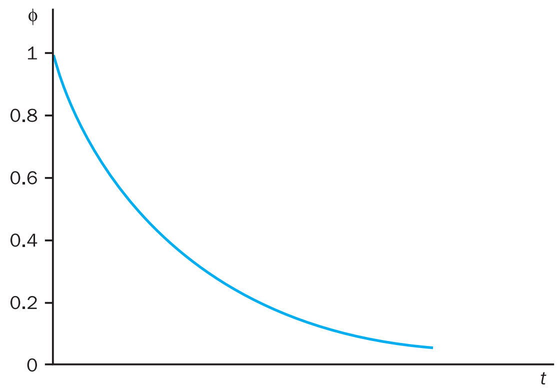


Figure 3.7 Utility weights with exponential discounting

rate is the discount rate ρ, as equations 3.16 and 3.17 indicate.

3.5.1.1 The arithmetic of discounting

Critics of the economic practice of exponential discounting point out that even at low rates of discount it discriminates strongly against the distant future and future generations. Table 3.3 illustrates. It shows the values of the weights φ_t for decades ahead for different values for the discount rate ρ. Using the intertemporal social welfare function which is equation 3.16, Table 3.3 shows, for example, that with $\rho = 0.01$ utility 10 years hence would be weighted by 0.9053 for a weight of unity on current utility. For $\rho = 0.04$ utility 10 years hence would be weighted by 0.6756. For 100 years hence the corresponding weights are 0.3697 and 0.0198.

To put this in some kind of perspective, we can think in terms of generations. Assume that on average children are born to parents at the age of 30. Then, using this kind of intertemporal social welfare function with $\rho = 0.01$ means that children's' future utility counts for about 75% as much as parents' current utility, while with $\rho = 0.04$ it counts for about 30% as much. Thinking about the future interests of grandchildren in this way means giving them about half the weight attached to our current interests for $\rho = 0.01$, or about 10% at $\rho = 0.04$. If we use $\rho = 0.04$ we weight our great grandchildren's interest at about 1% of our own – we are effectively ignoring the consequences for our great grandchildren in current decision making.

3.5.1.2 Why discount future utility?

What is the ethical basis for discounting future utility? How is the use of $\rho > 0$ morally justified? Basically there two schools of thought on this among economists, which can be usefully labelled 'descriptive' and 'prescriptive'.

Some economists, the descriptive school, argue that a positive rate of discount is required by the logic of the preference satisfaction variant of utilitarianism to which most economists adhere. Individuals as consumers are observed to exhibit positive time preference in that they require an incentive, in the form of the payment of interest, to postpone consumption, and hence utility, by saving. It follows from consumer sovereignty, it is argued, that in thinking about how society should make intertemporal choices, we should work with positive time preference in the form of $\rho > 0$.

Table 3.3 Values of φ_t for selected dates and discount rates

	Discount rate					
t	$\rho = 0.001$	$\rho = 0.01$	$\rho = 0.02$	$\rho = 0.03$	$\rho = 0.04$	
10	0.9901	0.9053	0.8204	0.7441	0.6756	
20	0.9802	0.8195	0.6730	0.5537	0.4564	
30	0.9705	0.7419	0.5521	0.4120	0.3083	Children
40	0.9608	0.6717	0.4529	0.3066	0.2083	
50	0.9513	0.6080	0.3715	0.2281	0.1407	
60	0.9418	0.5505	0.3048	0.1697	0.0951	Grandchildren
70	0.9324	0.4983	0.2500	0.1263	0.0642	
80	0.9232	0.4511	0.2051	0.0940	0.0434	
90	0.9140	0.4084	0.1683	0.0699	0.0293	Great Grandchildren
100	0.9049	0.3697	0.1380	0.0520	0.0198	

Other economists, the prescriptive school, argue that society should not adopt the preferences of individuals in this way. One version of this argument is a special case of a more general argument of the same nature. We noted above Sen's distinction between the individual's roles as consumer and citizen. This can be applied in the intertemporal context. As citizens exhibiting commitment toward others, future generations in this case, individuals would not necessarily wish to discount the future at the same rate as they do when considering the distribution of their own utility over time. Another version of the argument is specific to the intertemporal context. Pigou (1920), for example, argued that individuals suffer from a 'defective telescopic faculty', in that in taking decisions now they consistently underestimate future utility, with the degree of underestimation increasing with the degree of futurity. There is, the argument goes, no reason to carry this myopia into social decision making by discounting future utility.

Many people concerned about the environment argue, in effect, that in comparing utilities over time, the only ethically defensible position is that utilities attaching to each date should be treated equally, implying a zero rate of utility discounting, i.e. $\rho = 0$. Economists of the prescriptive school do not generally accept this. While accepting that people alive at different dates should have their utilities given equal regard, most economists from the prescriptive school would, nonetheless, argue for using $\rho > 0$ in equation 3.16 or 3.17.

The reason for this apparent inconsistency is as follows. Recall that economists take it that the only utility that counts is human utility. Most of the species that have existed in the history of earth are now extinct – extinction is the normal fate for a species. The argument is that the human species should not be assumed to be exempt from this fate. At any point in time, there is, it is argued, a very small, but non-zero, probability that the human species will go extinct. Candidate agents for such an event include meteorite impacts and new viruses. The probability of the human species existing diminishes the further ahead we look. Rather than try to guess when extinction will happen, and working with T in equation 3.15 set at that date, the argument is that we should take on board the declining probability of human existence by working with equation 3.16 or 3.17, where the time horizon is infinite, with $\rho > 0$ assigning smaller weights to dates at which existence is less likely. It can be shown that on reasonable assumptions, discounting justified by a declining probability of existence does imply exponentially declining utility weights.

This, prescriptive, argument for working with a positive constant discount rate is generally taken to imply low values for that rate, of the order of 0.001, 0.1%. Treating utilities at different dates equally, but discounting future utilities on account of a declining probability of existence, a value of 0.001 for ρ corresponds to a probability of human survival for 100 years of 0.905, that is a probability of 0.095 that humans will not survive 100 years.[3] Coming at things from the descriptive approach would typically involve a much higher number for ρ, of the order of 0.03, 3%. As Table 3.2 shows, which of these numbers one uses in thinking about the future matters a lot. For the former, my grandchild's utility counts for about 95% of mine. For the latter it counts for less than 25%.

We need to emphasise here that we have been, and in the rest of this chapter will be, discussing the discounting of utility, whereas much of the discussion of discounting in the economics literature is about the discounting of consumption. Clearly, with utility as a function of consumption, utility discounting and consumption discounting are connected. Failure to be clear about which is at issue can lead to confusion and error, and has often done so in the literature. We will look at the relationship between utility and consumption discounting in Chapter 11. Box 3.1 reports on an important investigation into the economics of climate change which highlighted the role of ethics in thinking about that problem, where the utility discount rate, together with the elasticity of marginal utility, played an important role.

[3] This value for ρ and the corresponding probabilities are taken from the Stern Review of the economics of climate change, Stern (2006). See also Pearce and Ulph (1995).

Box 3.1 Ethics and climate change in the Stern Review

The Stern Review (Stern 2006) is an economic analysis of the climate change problem, commissioned by the UK government and written by a team headed by Sir Nicholas (now Lord) Stern, a distinguished British economist known particularly for his work on welfare and public sector economics. The review was published in the late autumn of 2006. It is long, some 700 pages, and comprehensive. It, and lots of other material – papers by critics and replies by the review team for example – can be accessed at the Stern Review website, http://www.sternreview.org.uk. Here we are going to present a simplified account of just one aspect of the Stern Review, the role of ethics in the analysis. We will return to the review at various points in the rest of the book.

The 'climate change problem' is that human activities are putting greenhouse gases into the atmosphere at a faster rate than natural processes remove them from the atmosphere, so that atmospheric concentrations are increasing. This is a problem because the greenhouse gases trap infra-red radiation reflected by the earth's surface in the atmosphere, making it warmer than it would otherwise be, which warming leads to other climatic changes.[4] The most important greenhouse gas is carbon dioxide, CO_2.
In the Stern Review all greenhouse gases are considered. They are aggregated, using their warming effect per unit relative to that of CO_2, and the total is expressed in units which are carbon dioxide equivalent, CO_2e.

The principal conclusion of the Stern Review is that strong and early action to reduce greenhouse gas emissions makes good economic sense, with the benefits – in the form of avoided damage – of such action far outweighing the costs involved. This is in contrast to the position taken by most of the economists who had previously pronounced on the problem, and the review was criticised by a number of economists (see the review's website). Much of that criticism focused on the issues to be covered here.

The main arguments to the review's conclusion that are directly relevant here were as follows:

1. Doing nothing to reduce greenhouse gas emissions, i.e. 'Business as Usual' or BAU, would mean costs, on account of climate change damage, equivalent to an 10.9% reduction in global per capita consumption now and forever.
2. Stabilisation at 550 ppm (parts per million) CO_2e would reduce those costs to 1.1%.
3. The costs of action to stabilise at 550 ppm would be about 1% of gross world product.

So, BAU would involve large costs, which could be greatly reduced by stabilisation at 550 ppm, and the benefits of such stabilisation (i.e. the avoided costs, 10.9% – 1.1%) would considerably exceed the costs of achieving such stabilisation.

Currently concentrations are at 430 ppm CO_2e, and rising at 2 ppm per year. Stabilisation at 550 ppm CO_2e would require global greenhouse gas emissions to peak in the next 10 to 20 years, and by 2050 they would need to be around 25% lower than in 2000. Given population growth and economic growth, the reductions in emissions per capita and per unit of global GDP would be larger – on generally accepted projections for population and GDP growth, the global average for per capita emissions would need to fall to half of current levels and global CO_2e emissions per $ of global GDP would need to be one-quarter of current levels.

The Stern Review is very clear that thinking about what to do about climate change must pay explicit attention to ethics, and argues that many previous economic analyses did not pay enough attention to ethics. Because doing something about climate change involves taking action now to secure benefits, in the form of avoided climate change, which would continue far into the future, what we do about discounting is going to materially affect how much we think we should do now to abate greenhouse gas emissions.
We now explain how the Stern Review went about figuring the costs of climate change, or equivalently the benefits of reducing the amount of climate change.

[4] The climate change problem is also known as 'the greenhouse effect' or the 'enhanced greenhouse effect'. A short summary of the nature of the problem is given in Chapter 9 here, and see also chapter 13 in Common and Stagl (2005) for a brief overview of the science and the economics of the problem. The most recent comprehensive review of the science of climate change is the fourth assessment report of the Intergovernmental Panel on Climate Change published in 2007 – summaries of each of the Working Group reports are available at http://www.ipcc.ch/#.

Box 3.1 ***continued***

It used an 'Integrated Assessment Model' to generate a range of scenarios for greenhouse gas emissions and concentrations, and the arising damage due to climate change. The damage was represented as reduced consumption. It was assumed that

$$U(t) = \ln C(t)$$

so that the per capita utility function is iso-elastic with an elasticity of the marginal utility of consumption, η, equal to 1. Then, per capita utility is multiplied by the size of the global population, $N(t)$, and aggregated over time according to

$$W = \int_0^\infty N(t)\ln C(\mathrm{t})e^{-\rho t}$$

where ρ is the utility discount rate, for which the value 0.001 was used. Across all scenarios it is assumed that climate change damages cease in 2200. For each scenario

$$W = \int_0^\infty N(t)[(\ln C)e^{gt}]e^{-\rho t}$$

is solved for C. The solution C for a scenario is the current level of consumption which, growing at g, delivers the same amount of discounted utility as the scenario did. The growth rate g is the same across scenarios.

Thus, when the review says that for stabilisation at 550 ppm CO_2e the costs of climate change damage would be equivalent to a 1.1% reduction in global per capita consumption now and forever, it means that C for the 550 ppm stabilisation scenario is equal to 98.9% of the current level of per capita consumption. For the BAU scenario the damage costs were found to be similarly equivalent to an 11% reduction in per capita consumption now and forever.

Some economists criticised the Stern Review's chosen values for the parameters ρ and η, in both cases for being too low, so that damage caused by future climate change was given too much weight. That higher ρ means less weight on future costs (and benefits) is obvious. The origin of the relationship in the case of η is perhaps less so. The point is that, as shown in Table 3.1, higher η means less weight attached to gains/losses to the better off, in consumption terms. In the climate change problem as modelled by the Stern Review, and most economists, people in the future are better off than those alive now – it is assumed that economic growth will continue. In that case, increasing η means decreasing the weight attached to future as compared with current people, so reducing the significance of climate-change damages in current decision making.

The review authors responded to the criticism regarding values for ρ and η in two ways. First, they defended their choices, and second, they considered the implications of alternative, plausible, choices. As regards defending their choices, the review authors took a strongly prescriptive line on ρ, arguing that the only sound ethical basis for making it non-zero is the extinction argument, and that then 0.001 was the most reasonable value. As regards η, they conceded that higher values than 1 could be entertained, but argued that the implications of η > 2 rule out such values (see Table 3.1 here). The review authors were not, that is, prepared to go very far in the Rawlsian direction.

In one of their responses to their critics (Dietz *et al.*, 2007a, see Table 1), the review authors provided some sensitivity analysis on the BAU costs of climate change in regard to ρ and η. They report that increasing the value for ρ from 0.001 to 0.0015 reduces 10.9% to 3.1%, while increasing that for η from 1 to 2 reduces 10.9% to 3.4%. The review authors did not consider that this sensitivity, taken together with other analysis and data not discussed here, warranted any change to their main conclusion – that strong early action to reduce emissions, and hence stabilise concentrations, makes good economic sense. As noted above, we will come back to the Stern Review again in later chapters,

3.5.2 Optimal growth

Thus far in considering the utilitarian, and hence the economic (in the most part), approach to matters intertemporal, we have looked at things from the consumption and utility perspective, in which the big thing is impatience. For preference satisfaction utilitarianism, the reason for discounting is that individuals prefer consumption now to consumption in the future. There is another perspective to matters intertemporal, that of production, and the shifting of consumption, and hence utility, over time by the accumulation and use of capital. Just as economists take it as a major given, or stylised fact, that people are impatient, so they take it as a stylised fact that capital accumulation is productive in the sense that a unit of consumption forgone now for capital accumulation will pay off with more than one unit of future consumption. The study of optimal growth is the study of the interaction between impatience and productivity.

3.5.2.1 The basic model

The simplest exercise in optimal growth modelling is to find the path for consumption over time that results in

$$W = \int_{t=0}^{t=\infty} U(C_t)e^{-\rho t}\mathrm{d}t \tag{3.18}$$

taking the maximum value that is feasible, given the constraint that

$$\dot{K} = Q(K_t) - C_t \tag{3.19}$$

Equation 3.18 has the form of the utilitarian social welfare function introduced above. The only argument of the utility function is C_t, which is aggregate consumption at time t. We assume that population size is constant. It is also assumed that the, time invariant, relationship between society's utility and aggregate consumption is as shown in Figure 3.8 – utility increases with consumption at a decreasing rate. We are assuming that it makes sense to think in terms of society as a whole in the same way as for an individual.[5] With U_C for marginal utility, $U_C = \partial U/\partial C > 0$ and $U_{CC} = \partial^2 U/\partial C^2 < 0$. In the constraint 3.19, K stands for capital and $\dot{K}$ is the time derivative of K, i.e. the rate of investment. In this simple model, output is produced using just capital according to $Q(K_t)$.[6] The marginal product of capital is positive but declining, i.e.

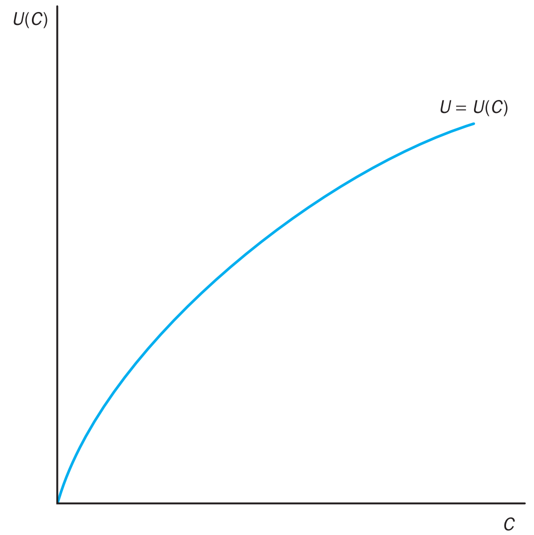


Figure 3.8 Utility as a function of aggregate consumption

$$Q_k = \frac{\partial Q_t}{\partial K_t} > 0 \text{ and } Q_{kk} = \frac{\partial^2 Q_t}{\partial K_t^2} < 0$$

Output can be either consumed or invested. The pay-off to investment follows from the positive marginal product of capital – a small amount of output invested today carries a cost in terms of current consumption of dC, but adds an amount larger than dC to future consumption possibilities.

This model is examined in some detail in Chapters 11 and 19. Here we just state one of the conditions that describes the optimal path for C_t, and briefly discuss its intuition and implications. The condition is

$$\frac{\dot{U}_C}{U_C} = \rho - Q_K \tag{3.20}$$

The left-hand side here is the proportional rate of change of marginal utility, and along the optimal consumption path this is equal to the difference between the utility discount rate and the marginal

[5] Chapter 11 looks, in an Appendix, at some of the aggregation issues that we are ignoring here.

[6] Having output depend on just capital input, rather than on inputs of capital and labour, simplifies the analysis and exposition without losing anything essential. Recall that we are assuming that the population size is constant.

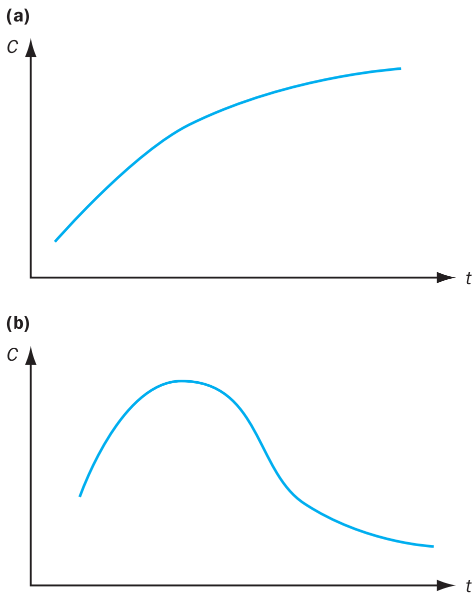


Figure 3.9 Optimal consumption growth paths

product of capital. The former is a constant parameter, while the marginal product of capital falls as the size of the capital stock increases.

Panel a of Figure 3.9 shows the nature of the optimal consumption path that equation 3.20 characterises given the standard assumptions. Initially the capital stock is small and its marginal product is high. So long as $Q_K > \rho$, the right-hand side of equation 3.20 is negative, and given diminishing marginal utility $\dot{U}_C/U_C < 0$ implies that consumption is increasing. This makes sense as the pay-off to deferring consumption, Q_K, is more than the cost of deferring it, ρ, so capital is accumulated, increasing output and consumption. As the size of the capital stock increases, so Q_K falls. For $Q_K = \rho$ the left-hand side of equation 3.20 is zero, so that growth and capital accumulation cease. Panel a of Figure 3.9 shows consumption going asymptotically to a level which is determined by the size of ρ and the properties of the production function.

Following an intertemporal consumption/savings plan derived from this model, those alive early would be saving for the benefit of those alive later, who will be richer, even though there is positive utility discounting. If planning were based on ρ = 0, savings at every point in time would be higher, and the accumulation of capital would continue until its marginal product was driven to zero. Given the productivity of savings and investment, zero discounting of utility would lead to poor early people doing lots of saving for the benefit of rich later people. On the other hand, with ρ relatively high, early people would do relatively little saving and accumulation, and the society might remain relatively poor despite the fact that it could become rich.

An important point here is the demonstration that the consequences of an ethical position depend on the circumstances in which it is acted upon. What appears to be an intrinsically sound ethical position may turn out in some circumstances to lead to outcomes that are not obviously sensible. Taken on its own, the position that we should work with ρ = 0, thus treating people alive at different times equally, seems ethically sound. However, if the world is like this simple model, following such a prescription would lead to what is not an obviously desirable outcome. Equally, ρ set at too high a value could also lead to what could be considered undesirable consequences. Adopting a consequentialist ethic means that we have to work out what consequences are, and they depend on the state of the world in which the ethic is adopted. The next sub-section further demonstrates this important lesson.

3.5.2.2 Optimal growth with non-renewable resources used in production

Now consider an exercise in optimal growth modelling that differs from the foregoing in just one respect – production uses inputs of a non-renewable natural resource as well as capital. The path for consumption, and hence savings and capital accumulation, is determined as that which maximises

$$W = \int_{t=0}^{t=\infty} U(C_t)e^{-\rho t}\mathrm{d}t \tag{3.21}$$

subject to the constraints

$$\dot{K} = Q(K_t, R_t) - C_t \tag{3.22a}$$

$$\dot{S} = -R_t \tag{3.22b}$$

$$\bar{S} = \int_{t=0}^{t=\infty} R_t \mathrm{d}t \tag{3.22c}$$

The first of the constraints says, as before, that output, Q, can either be used for consumption, C, or investment, $\dot{K}$. It differs from equation 3.19 in that the production of output now involves two inputs, capital, K, and some natural resource, R. The standard assumption is that the marginal product of the resource input is positive and diminishing. In equation 3.22b, S stands for stock, and this constraint says that the natural resource being used is non-renewable in that the stock size decreases by the amount used. $\bar{S}$ is the initial finite stock of the resource, and equation 3.22c says that total cumulative use of the resource cannot exceed the initial stock.

This problem will be considered in some detail in Chapter 14, and analysis of it and variants of it – such as for the case of a renewable resource – will take up much of Part IV. For present purposes we simply want to note that the optimal consumption path that this model produces (for a particular form for the production function, to be noted in the next section here) is as shown in panel b of Figure 3.9. Given $\rho > 0$, it is optimal for consumption first to increase, but eventually to start to decrease and to go asymptotically toward zero. Further, this is the case even when production technology is such that there is a constant level of consumption that could be maintained forever. It needs to be noted here that the production function $Q(K_t, R_t)$ may be such that constant consumption forever may simply be impossible: we return to this shortly.

As between panels a and b of Figure 3.9, the intertemporal social welfare functions are identical. What changes is the situation in regard to production. The morality of a positive rate of utility discount looks very different as between panels a and b in Figure 3.9.

3.5.3 Sustainability

The lesson to be drawn from panel b of Figure 3.9 is that if we live in a world where non-renewable resources are an essential input to production, then following utilitarianism with discounting makes people in the near future better off than we are, but makes people in the more distant future worse off, possibly very much worse off. In Chapter 2 we looked at the ways in which the economy and the natural environment are interdependent. We saw there that some people take it that the fact that production and consumption require inputs from the environment, and generate waste insertions into it, means that economic growth must come to an end – that there are 'limits to growth'. Indeed, we noted that there are those who consider that some limits have already been passed.

Many economists consider that the essential features of the starkest version of the dependence of production, and hence consumption, on the natural environment are captured by the model considered in the previous section. It seems obvious to many economists, that is, that if you want to consider economic limits set by the natural environment, the thing to do is to think about a world where production uses a non-renewable natural resource – such as oil – of which there exists a finite amount. To see why this is considered the 'canonical' model – the simplest one that captures the essential features – of the limits to growth problem, consider first an even simpler model. This is the 'cake-eating model'.

In this model, rather than the resource being used in production it is consumed directly, and it is the only thing that is consumed, and its consumption is necessary for existence. Given a finite stock of the resource, there is no positive level of consumption that can be maintained forever. Given a level of consumption x per year, and a stock of size $\bar{S}$, the stock is exhausted in $\bar{S}/x$ years, after which consumption goes to zero. In this model, the problem is not that consumption growth faces limits, but rather that there is no positive level of consumption that can be sustained indefinitely. The intertemporal distribution problem is now that of sharing out use of the resource over time. The problem with the obviously 'fair' solution of equal shares is that $\bar{S}$ is finite, so that the infinite planning horizon embodied in equation 3.21 would mean that the equal shares are of size zero – a finite cake cannot be divided into an infinite number of pieces. The intertemporal problem reduces to making the resource last as long as possible by consuming, at each point in time, as little as is possible consistent with survival.

3.5.3.1 Is sustainability feasible?

With the cake-eating model as background, the question for the model of production possibilities

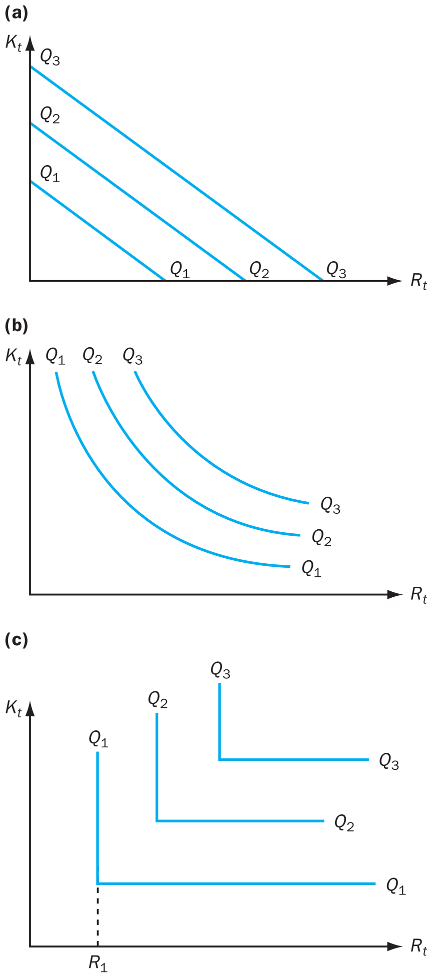


Figure 3.10 Production functions with capital and natural resource inputs

which is equations 3.22 is whether using the resource in production together with capital, rather than consuming it directly, makes it possible to sustain some positive level of consumption, and hence utility, indefinitely. Before we consider whether sustainability, a constant level of consumption/utility forever, is desirable, we must consider whether it is possible. The answer to this question turns on the possibilities for substitution between the resource and capital in production.

Figure 3.10 shows the isoquants for three specifications of the production function $Q_t = Q(K_t, R_t)$ that identify the possibilities. Panel a corresponds to:

$$Q_t = \alpha K_t + \beta R_t \tag{3.23}$$

In this case, the resource is non-essential in production. For $R_t = 0$, $Q_t = \alpha K_t$, and any level of output can be produced if there is enough capital. The use of a non-renewable resource in production does not, in this case, mean that sustainability as constant consumption is infeasible. Capital is a perfect substitute for the non-renewable resource.

Panel c of Figure 3.10 corresponds to:

$$Q_t = \min(\alpha K_t, \beta R_t) \tag{3.24}$$

In this case, Q_t is equal to whichever is the smaller of αK_t and βR_t. In panel c, given resource input R_1, for example, Q_1 is the maximum feasible output, however much capital input is used. In this case, the resource is essential in production, and substitution possibilities are non-existent. If there is no resource input, there is no output. Given the production function 3.24, the initial stock of the resource sets an upper limit to the amount that can be produced and consumed – total production over all time cannot exceed $\beta\bar{S}$. The situation here is essentially the same as with the cake-eating model discussed above.

With equation 3.23 as production function the intertemporal problem posed by the use of a non-renewable resource in production is trivial – constant consumption forever requires no special attention to the rate at which the resource is used. With equation 3.24 as production function there is no pattern of resource use over time that can make constant consumption forever feasible.

Panel b in Figure 3.10 shows the isoquants for a production function where capital is substitutable for the resource, but not perfectly. They are drawn for

$$Q_t = K_t^{\alpha} R_t^{\beta} \quad \text{with} \quad \alpha + \beta = 1 \tag{3.25}$$

which is a Cobb–Douglas production function with constant returns to scale. Clearly, if in equation 3.25 R_t is set at 0, then Q_t is 0 – the resource is essential in production. However, it can be shown (see Solow, 1974a) that given enough K, and $\alpha > \beta$, very high levels of output can be produced with very small levels of resource input, and that there exists a programme of capital accumulation such that R_t never actually becomes 0 (it goes asymptotically to zero) and consumption can be maintained constant forever.

3.5.3.2 Is sustainability optimal?

Many people take it as self-evident that it is wrong to act so as to make future generations worse off than we are. Having taken on board the economy–environment interdependence that we briefly described in Chapter 2, some people urge that we should conserve resources for future generations. Economists generally see the focus on resources *per se* as the wrong way of looking at the problem. As the Nobel Laureate economist Robert Solow put it, asking how much of a natural resource we should bequeath to our successors 'is a damagingly narrow way to pose the question'. According to Solow, our obligation to future generations 'refers to generalised productive capacity or, even wider, to certain standards of consumption/living possibilities' (Solow, 1986). What our successors will be interested in, Solow is in effect saying, is not the amount of 'oil' in the ground that they inherit from us, but rather whether they inherit the capability to do the things that we now do using 'oil'. They will be interested in the consumption opportunities that they inherit, not the stocks of resources that they inherit.

Clearly, it is implicit here that Solow believes that substitution possibilities in production are as shown in panel b of Figure 3.8, rather than in panel c. Most economists follow Solow in this regard – they do not believe that sustainability as constant consumption is a trivial problem, as it would be for panel a-type substitution possibilities, nor do they believe that it is simply infeasible, as it would be for panel c-type substitution possibilities. There is, then, a genuine intertemporal distribution problem to think about.

If we then adopt the standard intertemporal social welfare function with exponential utility discounting as in equation 3.21 and maximise subject to equations 3.22 where the production function is given the particular form of 3.25 with $\alpha > \beta$, we find that the optimal consumption path is that shown in panel b of Figure 3.9. Notwithstanding that constant consumption is feasible, it is not optimal. Given the stylised facts used to characterise production using a non-renewable resource, utilitarianism with exponential utility discounting leads to what many would regard as an 'immoral' outcome.

If we want a constant consumption sustainability outcome from utilitarianism we have to follow one of two routes. Either the intertemporal social welfare function has to be modified so as to lead to intertemporal equality, or additional constraints have to be incorporated into the maximisation problem.

An example of the first approach would be the adoption of a Rawlsian intertemporal social welfare function. Considering just two periods, 0 and 1, this would take the form:

$$W = \min(U_0, U_1) \tag{3.26}$$

With time periods, or generations, substituted for contemporary individuals, the previous discussion of this type of social welfare function applies. Solow (1974a) shows that generalising equation 3.26 for many periods and assuming production possibilities as per panel b of Figure 3.8, so that constant consumption forever is feasible, maximising such a welfare function leads to constant consumption forever as optimal.

An example of the second approach would be the adoption of some constraint on savings and investment behaviour, such as to bring about constant consumption as optimal. Hartwick (1977, 1978) showed that if a savings and investment constraint – now known as 'the Hartwick rule' – is added to the maximisation problem which is equations 3.21 and 3.22 above, then given the satisfaction of certain conditions, constant consumption emerges as the optimal programme. The Hartwick rule, the additional constraint, is that all of the rent arising from the extraction of the resource must be saved and invested. The conditions are that the production function takes the form of equation 3.25 with $\alpha > \beta$, and that the resource is extracted according to an intertemporally efficient programme. Intertemporal efficiency is discussed in general terms in Chapter 11, and for non-renewable resource extraction programmes in Chapter 14. As will be discussed in Chapter 19, in this model it turns out that following the Hartwick rule means that the total value of the economy's stock of capital together with its stock of the non-renewable resources is held constant over time – as the value of the remaining stock of the resource declines, so the value of the stock of capital increases in compensating amount.

3.5.3.3 Weak and strong sustainability

In some contributions to the sustainability literature a distinction is made between 'weak sustainability' and 'strong sustainability'. The use of adjectives to qualify the noun 'sustainability' suggests that two different kinds of sustainability are being distinguished. In fact, the distinction concerns the conditions that need to be met for the realisation of sustainability as constant consumption (or utility), rather than different conceptions or definitions of sustainability. 'Weak' is not a different kind of sustainability from 'strong'. Proponents of both weak and strong sustainability take constant utility to be what sustainability is. They differ over what is necessary for its realisation, and the difference is actually about possibilities for substitution in production. 'Weak sustainabilists' judge that these possibilities are essentially like those shown in panel b of Figure 3.10, while 'strong sustainabilists' judge that they are essentially like those shown in panel c of Figure 3.10.

The weak versus strong sustainability debate makes extensive use of the notion of 'natural capital', which we now explain. Production potential at any point in time depends on the stock of productive assets available for use. This stock can be classified into human labour and all other productive resources. Now define the term capital in a very broad sense, to include any economically useful stock, other than raw labour power. In this broad sense capital consists of:

(a) Natural capital: any naturally provided stock, such as aquifers and water systems, fertile land, crude oil and gas, forests, fisheries and other stocks of biomass, genetic material, and the earth's atmosphere itself. We have previously discussed, in Chapter 2, the services that the natural environment provides to the economy. Talking of 'natural capital' is a way of referring to the collectivity of the environmental assets from which all such services flow.
(b) Physical capital: plant, equipment, buildings and other infrastructure, accumulated by devoting part of current production to capital investment.
(c) Human capital: stocks of learned skills, embodied in particular individuals, which enhances the productive potential of those people.
(d) Intellectual capital: disembodied skills and knowledge and cultural practices. This stock is disembodied in that it does not reside in particular individuals, but is part of the culture of a society. It resides in books and other cultural artefacts and institutions, and is transmitted and developed through time by social learning processes. Sometimes, this kind of capital is called 'social' capital.

If human-made capital is defined to be the sum of physical, human and intellectual capital, then the total stock of capital stock can be seen as consisting of two parts: natural and human-made capital. The latter is sometimes referred to as reproducible capital. This is because it can be added to by human action, by diverting current output from consumption to saving and investment.

This way of classifying production inputs leads to writing the economy's production function in summary representative form as

$$Q = Q(L, K_N, K_H) \tag{3.27}$$

where L represents labour, K_N natural capital and K_H human-made capital. Within this framework, the difference between weak and strong sustainabilists turns on what they judge to be the extent of the substitution possibilities between K_N and K_H. The former consider that we should think about our obligations to the future assuming that K_H can substitute for K_N to the extent necessary to make constant consumption possible, the latter that we should assume that it is not. The former see the world as being like panel b of Figure 3.10, the latter like panel c.

The result is that proponents of strong sustainability argue that sustainability requires that the level of K_N is non-declining, while proponents of weak sustainability argue that it requires that it is the sum of K_N and K_H that must be non-declining. Clearly, going back to the previous sub-section, where we used K instead of K_H and R instead of K_N, Solow and Hartwick are weak sustainabilists. Most, but not all, economists are weak sustainabilists. Economists have tended to think about threats to sustainability as constant consumption mainly in terms of natural resource inputs to production and the possible exhaustion of the stocks of natural resources. It is in that context that their judgement that K_H can be substituted for K_N has to be understood. Historical experience does tend to support the idea that physical, human and intellectual capital accumulation can offset problems arising as stocks of natural resources are depleted. It is also true that there are many opportunities for substitution as between particulars of the general class of natural resources – bauxite for copper, for example.

It is in regard to the life-support and amenity services that natural capital provides, as discussed in Chapter 2, that there appears to be less ground for optimism about the extent to which human-made capital can be substituted for natural capital. As spacecraft have already demonstrated, it is possible to use K_H to provide necessary life-support services such as temperature control, breathable air etc., but only on a small scale. It has yet to be demonstrated, or even seriously argued, that human-made capital could replace natural capital in providing life-support services for several billions of humans. In regard to amenity services, some take the view that a lack of contact with the natural environment is bad for human well-being, and would argue that in this context we should regard the possibilities of K_H for K_N substitution as limited.

The strong and weak sustainability positions are statements about working assumptions regarding the broad picture, intended to highlight a fundamental question – that of substitutability. The question is not actually answerable at the level of K_H and K_N, which are each aggregations across many components. To say that K_H is substitutable for K_N is not to say one believes that there is no service that the environment provides that cannot be replaced by human activity, possibly based on some other environmental service. It is to say that one believes that there is a presumption that there is no environmental asset the services of which are irreplaceable in the sense depicted in panel c of Figure 3.8. This is the starting point for most economists in thinking, as utilitarians, about what our obligations to future people imply for current behaviour.

In regard to that, roughly speaking, weak sustainabilists say do not let the size of the total stock of capital fall, while strong sustainabilists say do not let the size of the natural capital stock fall. In order to do either, it is necessary to be able to measure the size of the natural capital stock. It is not a homogeneous thing, but consists of many qualitatively different components. How, then, does one define a single-valued measure of the natural capital stock? How do we add two lakes to one forest to get a single value for natural capital, for example? Anyone familiar with national income accounting will recognise this difficulty. National income accounts do have a single-valued measure of the quantity of output. To obtain this, weights are employed. For example, 2 cars plus 3 televisions would correspond to an output of 26 if we agreed to give each car a weight of 10 and each television a weight of 2 in the summation. For output of goods, an obvious weight to use is relative prices, and this is what is done in the national accounts.

But there are no obvious weights to use for aggregating individual items of natural capital. Prices do not exist for many items of natural capital and even where they do, there are many reasons why one would not be willing to accept them as correct reflections of 'true' values. If prices are to be used as weights, these prices will have to be imputed somehow or other. We discuss how this might be done in Chapter 12 (on valuation of environmental goods and services), and in Chapter 19 (on environmental accounting) we discuss aggregation using actual and imputed prices. Most economists would agree that no fully satisfactory method yet exists for valuing environmental resources, and certainly not all environmental resources have yet been valued. This means that a criterion which says that the total stock of natural capital should not be allowed to fall comes up against the fundamental problem that there is no satisfactory method of measuring the total stock of natural capital. Of course, if the stock of natural capital cannot be measured, then the total capital stock cannot be measured.

The basic point is that the kinds of analytical exercises that we have introduced here are to inform thinking rather than provide detailed prescriptions for complete solutions.

3.5.3.4 Ecologists on sustainability

As we noted in Chapters 1 and 2, ecologists are also concerned with the interdependence of economy and environment. Ecologists also talk about 'sustainability' as a problem arising from that interdependence. We finish this chapter by looking very briefly at the way that ecologists conceptualise this problem. In so far as their arguments can be cast within the strong/weak sustainability framework, most ecologists are strong sustainabilists – in effect, they judge the possibilities for substituting K_H for K_N to be rather limited. That is not, however, how they come at the problem.

Ecological science looks at its subject matter within a systems perspective. The whole system – the biosphere – consists of an interlocking set of ecological subsystems or ecosystems. Systems analysts are concerned with organisational characteristics and structure, and with systems dynamics – processes of evolution and change.

Ecologists look at sustainability from the point of view of an ecological system of which humans are just one part. Sustainability is assessed in terms of the extent to which the prevailing structure and properties of the ecosystem can be maintained. Human interests are not regarded as paramount; rather, they are identified with the continuing existence and functioning of the biosphere in a form more or less similar to that which exists at present. Thus:

> Sustainability is a relationship between human economic systems and larger dynamic, but normally slower-changing, ecological systems in which 1) human life can continue indefinitely, 2) human individuals can flourish, and 3) human cultures can develop; but in which effects of human activities remain within bounds, so as not to destroy the diversity, complexity, and function of the ecological life support system.
>
> *(Costanza et al., 1991, p. 8)*

Ecological views are often more human-centred, anthropocentric, than is made explicit in their advocacy. There is generally a presumption, often implicit, that the present system structure, including the important place in it occupied by humans, is to be preferred to others. To confirm that this is so, consider the attitude an ecologist might take to the threat of global warming. If large-scale global warming were to occur, there is a high probability that major ecosystem change would occur. The biosphere would not cease to operate – it would just operate in a different way. We guess that nearly all ecologists would take a stand against global warming, and most would do so on the grounds that human life is more threatened in a changed ecosystem than in the present one. Some non-human species would be more favoured in a biosphere operating in substantially different ways from that which it currently does.

In this spirit, Common and Perrings (1992) argue that ecological sustainability is a prerequisite for the sustainability of the joint environment–economy system, and that ecological sustainability requires resilience. The concept of resilience was introduced in Chapter 2. We can say that an ecosystem is resilient if it maintains its functional integrity in the face of exogenous disturbance. Common and Perrings show that satisfying the conditions for intertemporal economic efficiency (which are set out in detail in Chapter 11 here) and following the Hartwick rule is neither necessary nor sufficient for sustainability as resilience. An economy–environment system could, that is, be sustainable in their sense without satisfying those conditions, and, on the other hand, an economy which satisfied those conditions could be unsustainable in their sense. Basically, the problem is that the economic conditions reflect individuals' preferences – they derive from consumer sovereignty – and there is no reason to suppose that those preferences reflect the requirements of resilience.

As noted in Chapter 2, while we are able to observe whether a system is resilient after a disturbance has taken place, *ex ante* we cannot know whether a system will be resilient in the face of future shocks that it will be subject to. Further, we do not know what form those future shocks will take. We do know that a system could be resilient in the face of a shock of one sort, but not in the face of one of a different sort. Uncertainty pervades the behaviour of ecological systems, ensuring that we cannot know in advance whether some system is or is not resilient.

Paralleling the economic idea of monitoring the stock of capital as a sustainability indicator, some ecologists have suggested that some environmental indicators are useful as monitoring devices, and can be used to make inferences about potential changes in the degree of resilience of ecosystems in which we are interested. Schaeffer *et al.* (1988), for example, propose a set of indicators, including:

- changes in the number of native species;
- changes in standing crop biomass;
- changes in mineral micronutrient stocks;
- changes in the mechanisms of and capacity for damping oscillations.

Suggestive as these and other indicators might be, none can ever be a completely reliable instrument in the sense that a satisfactory rating can be taken as a guarantee of resilience. We return to the question of sustainability indicators, mainly from an economic perspective, in Chapter 19.

Summary

Policy objectives

Economists make recommendations concerning environmental policy objectives, such as, for example, the level of pollution to be allowed. Doing this necessarily involves the adoption of some ethical criteria, since there are always feasible options from which a selection must be made.

Utilitarianism

Most economists derive their recommendations from welfare economics, the ethical basis for which is a form of utilitarianism where the criterion of what is good for a human individual is that individual's own tastes. Many of those who are concerned about the natural environment have different ethical positions. Some want, for example, to confer moral standing on non-human individuals.

Discounting

Should future humans be treated equally to current humans? The descriptive approach to this question would say 'no', use a positive utility discount rate, because that is what we observe people to do. The prescriptive approach bases the use of a positive rate on the decreasing probability of human existence with greater futurity.

Sustainability

Planning for the future with a positive utility discount rate may entail that future humans are worse off than those alive now. Whether it is feasible for the future to be as well off depends on the possibilities for substitution in production.

Further reading

A good introduction to ethics, including environmental applications, may be found in Singer (1993). Sen (1987) looks at ethics in relation to economics. Beauchamp and Bowie (1988) give a good presentation, especially in chapters 1 and 9; the book contains an interesting analysis of the business implications of ethical principles. Penz (1986) and Scitovsky (1986) consider consumer sovereignty and its problems.

Kneese and Schulze (1985) and Glasser (1999) are survey articles dealing specifically with ethics in relation to environmental economics and policy analysis, both of which provide extensive references to the literature. The journal *Environmental Values* aims to bring together contributions from philosophy, law, economics and other disciplines concerning the ethical basis for our treatment of the natural environment. The February/March 1998 issue (Vol. 24, Nos 2,3) of *Ecological Economics* was a special issue on 'Economics, ethics and environment'. Heal (2005) is an overview of ethics and utility discounting.

Bruni and Porta (2005) is a collection of papers which are well referenced and provide a good overview of and introduction to the happiness literature: see also Frey and Stutzer (2002). The World Data Base of Happiness, http://www1.eur.nl/fsw/happiness/, is very useful. As well as data from surveys of SWB, it provides a comprehensive happiness bibliography, a directory of happiness researchers, and data on other variables. Layard (2005a), which appears in Bruni and Porta (2005), is an exercise in drawing out the implications for welfare economics of results from the happiness literature. Layard (2005b) is a non-technical introduction to happiness research. Of those economists aware of the happiness research, not all are convinced that it has much relevance for economics: see Johns and Ormerod (2007). White (2006) provides a history of the treatment of the idea of happiness in philosophy: see also McMahon (2006). Nettle (2005) is a short introduction to the psychology of happiness, with an emphasis on evolutionary psychology.

References on intertemporal welfare economics will be provided with Chapter 11. The Stern Review (Stern, 2006) can be downloaded from http://www.sternreview.org.uk, as can many of the papers criticising it, and the review team's responses (Dietz *et al.*, 2007a, 2007b). Ackerman (2007), downloadable at http://www.foe.co.uk/resource/reports/debate_climate_econs.pdf, overviews and evaluates the exchanges between the review team and critics. Spash (2007b) is a critical review of the Stern Review, focusing largely on the ethical aspects.

Seminal economic contributions on sustainability in the framework of the neoclassical growth model are to be found in Solow (1974b, 1986) and Hartwick (1977, 1978); further references in this area will be given in Chapter 19. Farmer and Randall (1997) present an overlapping generations model in which sustainability issues are examined. van den Bergh and Hofkes (1999) review economic models of sustainable development, while Faucheux *et al.* (1996) contains a number of models representing differing disciplinary approaches. Good general surveys are presented in Barbier (1989a), Klassen and Opschoor (1991), Markandya and Richardson (1992), Toman *et al.* (1993) and Neumayer (1999), which has a very comprehensive bibliography. The ecological economics approach to sustainability is explored in various contributions to Köhn *et al.* (1999): see also Pearce (1987), Costanza (1991), Common and Perrings (1992) and Söderbaum (2000). Page (1997) compares two approaches to the problem of achieving the goals of sustainability and intergenerational efficiency.

Historically interesting contributions from a more ecological perspective are: Boulding (1966) and Daly (1974, 1977, 1987). An interesting assessment of the contribution of scientific understanding to the debate is to be found in Ludwig *et al.* (1993).

Discussion questions

1. We argued in the text that Rawls's Difference Principle asserts that it is only just to have an unequal distribution of wealth if all persons benefit from that allocation, relative to the situation of an equal distribution. But we also argued that the total level of utility attainable might depend on the distribution of wealth, as utility could be higher in an unequal position if incentive effects enhance productive efficiency. Discuss the implications of these comments for a morally just distribution of resources within and between countries.
2. In discussing the work of Robert Nozick, it was argued that libertarian ethics have been adopted most enthusiastically by those who believe in a limited role for government. But we also noted that it is by no means clear that a completely laissez-faire approach is necessarily implied. We noted three difficult issues arising in connection with the principle of just acquisition:

   What should government do about unjust holdings?
   How are open access or common property resources to be dealt with?

How do external effects and public goods relate to the concept of just acquisition? Sketch out reasoned answers to these three questions.

3. If society deemed it to be correct that some animals or plants have intrinsic rights (such as rights to be left undisturbed or rights to be reasonably protected), then such rights can be protected by imposing them as constraints on what is legitimate human behaviour. Do humans appear to regard whales as having intrinsic rights, and if so, what rights are these? In what ways, if at all, do humans defend these rights by imposing constraints on human behaviour?

4. A river tumbles through forested ravines and rocky gorges towards the sea. The state hydro-electricity commission sees the falling water as untapped energy. Building a dam across one of the gorges would provide three years of employment for a thousand people, and provide longer-term employment for 20 or 30. The dam would store enough water to ensure that the state could economically meet its energy needs for the next decade. This would encourage the establishment of energy-intensive industry, thus further contributing to employment and economic growth.

   The rough terrain of the river valley makes it accessible only to the reasonably fit, but it is nevertheless a favoured spot for bush-walking. The river itself attracts the more daring whitewater rafters. Deep in the sheltered valleys are stands of rare Huon Pine, many of the trees being over a thousand years old. The valleys and gorges are home to many birds and animals, including an endangered species of marsupial mouse that has seldom been found outside the valley. There may be other rare plants and animals as well, but no one knows, for scientists are yet to investigate the region fully.

   (Singer, 1993, p. 264)

   Peter Singer's discussion of ethics and the environment begins with this scenario. His description is loosely based on a proposed dam on the Franklin River in Tasmania. Singer notes that this is an example of a situation in which we must choose between very different sets of values. Please answer the following question, as put by Singer: Should the dam be built?

5. Given that the question of the substitution possibilities as between human-made and natural capital is so important, how can the fact that there is no generally agreed answer be explained?

Problems

1. Suppose that one believed that each generation should have the same level of well-being as every other one. Demonstrate that we could not ensure the attainment of this merely by the choice of a particular discount rate, zero or otherwise.

2. Prove that, under the assumption of diminishing marginal utility, the linear indifference curves in utility space in Figure 3.1 map into indifference curves that are convex from below in commodity space, as illustrated in Figure 3.2.

3. Demonstrate that an unequal distribution of goods at a welfare maximum may occur when the weights attached to individual utilities are not equal, and/or when individuals have different utility functions.

4. Find the marginal and average products of K and R for the Cobb–Douglas production function 3.22 with $\alpha + \beta = 1$. How do your results relate to the feasibility of indefinite constant consumption despite the fact that $Q = 0$ for $R = 0$?

CHAPTER 4 Welfare economics and the environment

Welfare economics is the branch of economic theory which has investigated the nature of the policy recommendations that the economist is entitled to make.

Baumol (1977), p. 496

Learning objectives

In this chapter you will

- learn about the concepts of efficiency and optimality in allocation
- derive the conditions that are necessary for the realisation of an efficient allocation
- find out about the circumstances in which a system of markets will allocate efficiently
- learn about market failure and the basis for government intervention to correct it
- find out what a public good is, and how to determine how much of it the government should supply
- learn about pollution as an external effect, and the means for dealing with pollution problems of different kinds
- encounter the second best problem

Introduction

When economists consider policy questions relating to the environment they draw upon the basic results of welfare economics. The purpose of this chapter is to consider those results from welfare economics that are most relevant to environmental policy problems. Efficiency and optimality are the two basic concepts of welfare economics, and this chapter explains these concepts as they relate to problems of allocation. There are two classes of allocation problem, static and intertemporal. Efficiency and optimality are central to both. In this chapter we confine attention to the static problem – the allocation of inputs across firms and of outputs across individuals at a point in time. The intertemporal problem – allocation over time – is dealt with in Chapter 11. If you have previously studied a course in welfare economics, you should be able to read through the material of this chapter rather quickly. If not, the chapter will fill that gap.

There are three parts to this chapter. The first states and explains the conditions required for an allocation to be (a) efficient and (b) optimal. These conditions are derived without regard to any particular institutional setting. In the second part of the chapter, we consider how an efficient allocation would be brought about in a market economy characterised by particular institutions. The third part of the chapter looks at the matter of 'market failure' – situations where the institutional conditions required for the operation of pure market forces to achieve efficiency in allocation are not met – in relation to the environment.

PART I EFFICIENCY AND OPTIMALITY

In this part, and the next, of this chapter we will, following the usage in the welfare economics literature, use ‘resources’ to refer generally to inputs to production rather than specifically to extractions from the natural environment for use in production. In fact, in these parts of the chapter, when we talk about resources, or ‘productive resources’, we will have in mind, as we will often make explicit, inputs of capital and labour to production.

At any point in time, an economy will have access to particular quantities of productive resources. Individuals have preferences about the various goods that it is feasible to produce using the available resources. An allocation of resources, or just an allocation, describes what goods are produced and in what quantities they are produced, which combinations of resource inputs are used in producing those goods, and how the outputs of those goods are distributed between persons.

In this section, and the next, we make two assumptions that will be relaxed in the third part of this chapter. First, that no externalities exist in either consumption or production; roughly speaking, this means that consumption and production activities do not have un-intended and un-compensated effects upon others. Second, that all produced goods and services are private (not public) goods; roughly speaking, this means that all outputs have characteristics that permit of exclusive individual consumption on the part of the owner.

In the interests of simplicity, but with no loss of generality, we strip the problem down to its barest essentials. Our economy consists of two persons (A and B); two goods (X and Y) are produced; and production of each good uses two inputs (K for capital and L for labour) each of which is available in a fixed quantity.

Let U denote an individual’s total utility, which depends only on the quantities of the two goods that he or she consumes. Then we can write the utility functions for A and B in the form shown in equations 4.1.

$$
\begin{aligned}
U^{A} &= U^{A}(X^{A}, Y^{A}) \\
U^{B} &= U^{B}(X^{B}, Y^{B})
\end{aligned}
\tag{4.1}
$$

The total utility enjoyed by individual A, denoted U^{A}, depends upon the quantities he or she consumes of the two goods, X^{A} and Y^{A}. An equivalent statement can be made about B’s utility.

Next, we suppose that the quantity produced of good X depends only on the quantities of the two inputs K and L used in producing X, and the quantity produced of good Y depends only on the quantities of the two inputs K and L used in producing Y. Thus, we can write the two production functions in the form shown in 4.2:

$$
\begin{aligned}
X &= X(K^{X}, L^{X}) \\
Y &= Y(K^{Y}, L^{Y})
\end{aligned}
\tag{4.2}
$$

Each production function specifies how the output level varies as the amounts of the two inputs are varied. In doing that, it assumes technical efficiency in production. The production function describes, that is, how output depends on input combinations, given that inputs are not simply wasted. Consider a particular input combination K^{X} and L^{X} with X given by the production function. Technical efficiency means that in order to produce more of X it is necessary to use more of K^{X} and/or L^{X}.

The marginal utility that A derives from the consumption of good X is denoted U^{A}_{X}; that is, $U^{A}_{X} = \partial U^{A}/\partial X^{A}$. Equivalent notation applies for the other three marginal utilities. The marginal product of the input L in the production of good Y is denoted MP^{Y}_{L}; that is, $MP^{Y}_{L} = \partial Y/\partial L^{Y}$. Equivalent notation applies for the other three marginal products.

The marginal rate of utility substitution for A is the rate at which X can be substituted for Y at the margin, or vice versa, while holding the level of A’s utility constant. It varies with the levels of consumption of X and Y and is given by the slope of the indifference curve. We denote A’s marginal rate of substitution as $MRUS^{A}$, and similarly for B.

The marginal rate of technical substitution as between K and L in the production of X is the rate at which K can be substituted for L at the margin, or vice versa, while holding the level output of X constant. It varies with the input levels for K and L and is given by the slope of the isoquant. We denote the marginal rate of substitution in the production of X as $MRTS_{X}$, and similarly for Y.

The marginal rates of transformation for the commodities X and Y are the rates at which the output of one can be transformed into the other by marginally shifting capital or labour from one line of production to the other. Thus, MRT_{L} is the increase in the output of Y obtained by shifting a small, strictly an infinitesimally small, amount of labour from use in the production of X to use in the production of Y, or vice versa. Similarly, MRT_K is the increase in the output of Y obtained by shifting a small, strictly an infinitesimally small, amount of capital from use in the production of X to use in the production of Y, or vice versa.

With this notation we can now state, and provide intuitive explanations for, the conditions that characterise efficient and optimal allocations. Appendix 4.1 uses the calculus of constrained optimisation (which was reviewed in Appendix 3.1) to derive these conditions formally.

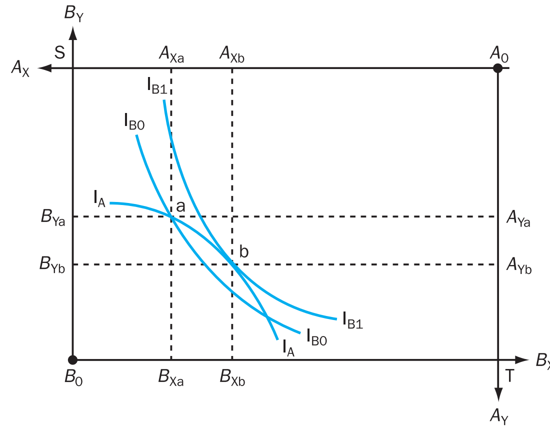


Figure 4.1 Efficiency in consumption

4.1 Economic efficiency

An allocation of resources is said to be efficient if it is not possible to make one or more persons better off without making at least one other person worse off. Conversely, an allocation is inefficient if it is possible to improve someone's position without worsening the position of anyone else. A gain by one or more persons without anyone else suffering is known as a Pareto improvement. When all such gains have been made, the resulting allocation is sometimes referred to as Pareto optimal, or Pareto efficient. A state in which there is no possibility of Pareto improvements is sometimes referred to as being allocatively efficient, rather than just efficient, so as to differentiate the question of efficiency in allocation from the matter of technical efficiency in production.

Efficiency in allocation requires that three efficiency conditions are fulfilled – efficiency in consumption, efficiency in production, and product-mix efficiency.

4.1.1 Efficiency in consumption

Consumption efficiency requires that the marginal rates of utility substitution for the two individuals are equal:

$$MRUS^A = MRUS^B \tag{4.3}$$

If this condition were not satisfied, it would be possible to rearrange the allocation as between A and B of whatever is being produced so as to make one better off without making the other worse off. Figure 4.1 shows what is involved by considering possible allocations of fixed amounts of X and Y between A and B.[1] The top right-hand corner, labelled A_0, refers to the situation where A gets nothing of the available X or Y, and B gets all of both commodities. The bottom left-hand corner, B_0, refers to the situation where B gets nothing and A gets everything. Starting from A_0, moving horizontally left measures A's consumption of X, and moving vertically downwards measures A's consumption of Y. As A's consumption of a commodity increases, so B's must decrease. Starting from B_0, moving horizontally right measures B's consumption of X, and moving vertically upwards measures B's consumption of Y. Any allocation of X and Y as between A and B is uniquely identified by a point in the box SA_0TB_0. At the point a, for example, A is consuming A_0A_{Xa} of X and A_0A_{Ya} of Y, and B is consuming B_0B_{Xa} of X and B_0B_{Ya} of Y.

The point a is shown as lying on I_AI_A which is an indifference curve for individual A. I_AI_A may look odd for an indifference curve, but remember that it is drawn with reference to the origin for A which is

[1] This figure is an 'Edgeworth box'.

the point A_0. Also shown are two indifference curves for B, $I_{B0}I_{B0}$ and $I_{B1}I_{B1}$. Consider a reallocation as between A and B, starting from point a and moving along I_AI_A, such that A is giving up *X* and gaining *Y*, while B is gaining *X* and giving up *Y*. Initially, this means increasing utility for B, movement onto a higher indifference curve, and constant utility for A. However, beyond point b any further such reallocations will involve decreasing utility for B. Point b identifies a situation where it is not possible to make individual B better off while maintaining A's utility constant – it represents an efficient allocation of the given amounts of *X* and *Y* as between A and B. At b, the slopes of I_AI_A and $I_{B1}I_{B1}$ are equal – A and B have equal marginal rates of utility substitution.

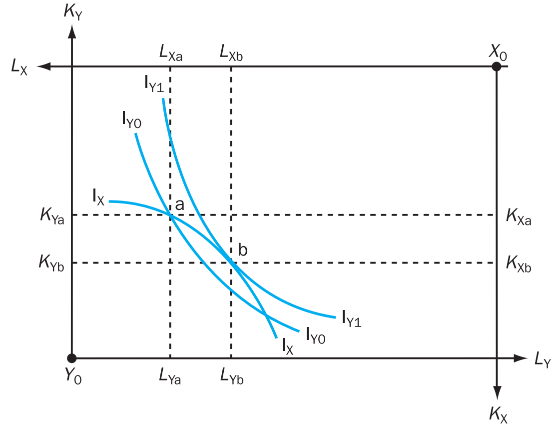


Figure 4.2 Efficiency in production

4.1.2 Efficiency in production

Turning now to the production side of the economy, recall that we are considering an economy with two inputs, *L* and *K*, which can be used (via the production functions of equations 4.2) to produce the goods *X* and *Y*. Efficiency in production requires that the marginal rate of technical substitution be the same in the production of both commodities. That is

$$MRTS_X = MRTS_Y \tag{4.4}$$

If this condition were not satisfied, it would be possible to reallocate inputs to production so as to produce more of one of the commodities without producing less of the other. Figure 4.2 shows why this condition is necessary. It is constructed in a similar manner to Figure 4.1, but points in the box refer to allocations of capital and labour to the production of the two commodities rather than to allocations of the commodities between individuals.[2] At X_0 no capital or labour is devoted to the production of commodity *X* – all of both resources is used in the production of *Y*. Moving horizontally to the left from X_0 measures increasing use of labour in the production of *X*, moving vertically down from X_0 measures increasing use of capital in the production of *X*. The corresponding variations in the use of inputs in the production of *Y* – any increase/decrease in use for *X* production must involve a decrease/increase in use for *Y* production – are measured in the opposite directions starting from origin Y_0.

I_XI_X is an isoquant for the production of commodity *X*. Consider movements along it to the southeast from point a, so that in the production of *X* capital is being substituted for labour holding output constant. Correspondingly, given the full employment of the resources available to the economy, labour is being substituted for capital in the production of *Y*. $I_{Y0}I_{Y0}$ and $I_{Y1}I_{Y1}$ are isoquants for the production of *Y*. Moving along I_XI_X from a toward b means moving onto a higher isoquant for *Y* – more *Y* is being produced with the production of *X* constant. Movement along I_XI_X beyond point b will mean moving back to a lower isoquant for *Y*. The point b identifies the highest level of production of *Y* that is possible, given that the production of *X* is held at the level corresponding to I_XI_X and that there are fixed amounts of capital and labour to be allocated as between production of the two commodities. At point b the slopes of the isoquants in each line of production are equal – the marginal rates of technical substitution are equal. If these rates are not equal, then clearly it would be possible to reallocate inputs as between the two lines of production so as to produce more of one commodity without producing any less of the other.

[2] Appendix 4.1 establishes that all firms producing a given commodity are required to operate with the same marginal rate of technical substitution. Here we are assuming that one firm produces all of each commodity.

Box 4.1 Productive inefficiency in ocean fisheries

The total world marine fish catch increased steadily from the 1950s through to the late 1980s, rising by 32% between the periods 1976–1978 and 1986–1988 (UNEP, 1991). However, the rate of increase was slowing toward the end of this period, and the early 1990s witnessed downturns in global harvests. The harvest size increased again in the mid-1990s, the 1996 harvest was a new peak, and then levelled off again in the late 1990s. It is estimated that the global maximum sustainable harvest is about 10% larger than harvest size in the late 1990s.

The steady increase in total catch until 1989 masked significant changes in the composition of that catch; as larger, higher-valued stocks became depleted, effort was redirected to smaller-sized and lower-valued species. This does sometimes allow depleted stocks to recover, as happened with North Atlantic herring, which recovered in the mid-1980s after being overfished in the late 1970s. However, many fishery scientists believe that these cycles of recovery have been modified, and that species dominance has shifted permanently towards smaller species.

Rising catch levels have put great pressure on some fisheries, particularly those in coastal areas, but also including some pelagic fisheries. Among the species whose catch declined over the period 1976–1988 are Atlantic cod and herring, haddock, South African pilchard and Peruvian anchovy. Falls in catches of these species have been compensated for by much-increased harvests of other species, including Japanese pilchard in the north-west Pacific.

Where do inefficiencies enter into this picture? We can answer this question in two ways. Firstly, a strong argument can be made to the effect that the total amount of resources devote to marine fishing is excessive, probably massively so. We shall defer giving evidence to support this claim until Chapter 17 (on renewable resources), but you will see there that a smaller total fishing fleet would be able to catch at least as many fish as the present fleet does. Furthermore, if fishing effort were temporarily reduced so that stocks were allowed to recover, a greater steady-state harvest would be possible, even with a far smaller world fleet of fishing vessels. There is clearly an inefficiency here.

A second insight into inefficiency in marine fishing can be gained by recognising that two important forms of negative external effect operate in marine fisheries, both largely attributable to the fact that marine fisheries are predominantly open-access resources. One type is a so-called crowding externality, arising from the fact that each boat's harvesting effort increases the fishing costs that others must bear. The second type may be called an intertemporal externality: as fisheries are often subject to very weak (or even zero) access restrictions, no individual fisherman has an incentive to conserve stocks for the future, even if all would benefit if the decision were taken jointly.

As the concepts of externalities and open access will be explained and analysed in the third part of this chapter, and applied to fisheries in Chapter 17, we shall not explain these ideas any further now. Suffice it to say that production in market economies will, in general, be inefficient in the presence of external effects.

Sources: WRI (2000), WRI website, FAO website

4.1.3 Product-mix efficiency

The final condition necessary for economic efficiency is product-mix efficiency. This requires that

$$MRT_L = MRT_K = MRUS^A = MRUS^B \tag{4.5}$$

This condition can be understood using Figure 4.3. Given that equation 4.3 holds, so that the two individuals have equal marginal rates of utility substitution and $MRUS^A = MRUS^B$, we can proceed as if they had the same utility functions, for which II in Figure 4.3 is an indifference curve with slope MRUS. The individuals do not, of course, actually have the same utility functions. But, given the equality of the MRUS, their indifference curves have the same slope at an allocation that satisfies the consumption efficiency condition, so we can simplify, without any real loss, by assuming the same utility functions and drawing a single indifference curve that refers to all consumers. Given that equation 4.4 holds, when we think about the rate at which the economy can trade-off production of X for Y and vice versa, it does not matter whether the changed composition of consumption is realised by switching labour or capital

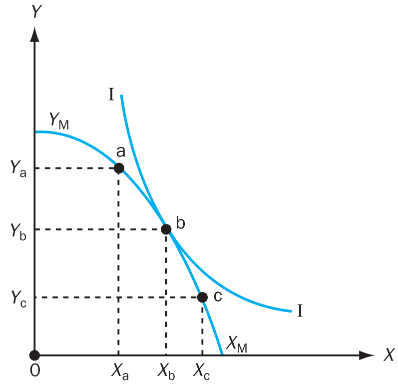


Figure 4.3 Product-mix efficiency

between the two lines of production. Consequently, in Figure 4.3 we show a single production possibility frontier, $Y_M X_M$, showing the output combinations that the economy could produce using all of its available resources. The slope of $Y_M X_M$ is MRT.

In Figure 4.3 the point a must be on a lower indifference curve than II. Moving along $Y_M X_M$ from a toward point b must mean shifting to a point on a higher indifference curve. The same goes for movement along $Y_M X_M$ from c toward b. On the other hand, moving away from b, in the direction of either a or c, must mean moving to a point on a lower indifference curve. We conclude that a point like b, where the slopes of the indifference curve and the production possibility frontier are equal, corresponds to a product mix – output levels for X and Y – such that the utility of the representative individual is maximised, given the resources available to the economy and the terms on which they can be used to produce commodities. We conclude, that is, that the equality of MRUS and MRT is necessary for efficiency in allocation. At a combination of X and Y where this condition does not hold, some adjustment in the levels of X and Y is possible which would make the representative individual better off.

An economy attains a fully efficient static allocation of resources if the conditions given by equations 4.3, 4.4 and 4.5 are satisfied simultaneously. Moreover, it does not matter that we have been dealing with an economy with just two persons and two goods. The results readily generalise to economies with many inputs, many goods and many individuals. The only difference will be that the three efficiency conditions will have to hold for each possible pairwise comparison that one could make, and so would be far more tedious to write out.

4.2 An efficient allocation of resources is not unique

For an economy with given quantities of available resources, and given production functions and utility functions, there will be many efficient allocations of resources. The criterion of efficiency in allocation does not, that is, serve to identify a particular allocation.

To see this, suppose first that the quantities of X and Y to be produced are somehow given and fixed. We are then interested in how the given quantities of X and Y are allocated as between A and B, and the criterion of allocative efficiency says that this should be such that A/B cannot be made better off except by making B/A worse off. This was what we considered in Figure 4.1 to derive equation 4.3, which says that an efficient allocation of fixed quantities of X and Y will be such that the slopes of the indifference curves for A and B will be the same. In Figure 4.1 we showed just one indifference curve for A and two for B. But, these are just a small subset of the indifference curves for each individual that fill the box SA_0TB_0. In Figure 4.4 we show a larger subset for each individual. Clearly, there will be a whole family of points, like b in Figure 4.1, at which the slopes of the indifference curves for A and B are equal, at which they have equal marginal rates of utility substitution. At any point along CC in Figure 4.4, the consumption efficiency condition is satisfied. In fact, for given available quantities of X and Y there are an indefinitely large number of allocations as between A and B that satisfy $MRUS^A = MRUS^B$.

Now consider the efficiency in production condition, and Figure 4.2. Here we are looking at variations in the amounts of X and Y that are produced. Clearly, in the same way as for Figures 4.1 and 4.4, we could introduce larger subsets of all the possible isoquants for the production of X and Y to show that there are many X and Y combinations that satisfy equation 4.4,

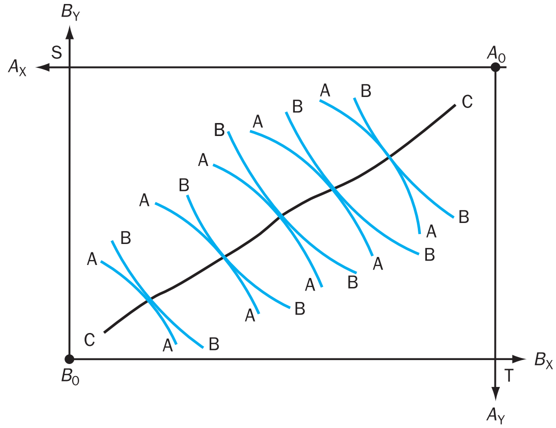


Figure 4.4 The set of allocations for consumption efficiency

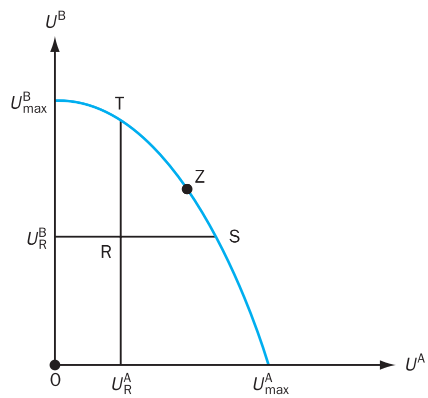


Figure 4.5 The utility possibility frontier

combinations representing uses of capital and labour in each line of production such that the slopes of the isoquants are equal, $MRTS_X = MRTS_Y$.

So, there are many combinations of X and Y output levels that are consistent with allocative efficiency, and for any particular combination there are many allocations as between A and B that are consistent with allocative efficiency. These two considerations can be brought together in a single diagram, as in Figure 4.5, where the horizontal axis measures A's utility and the vertical B's. Consider a particular allocation of capital and labour as between X and Y production which implies particular output levels for X and Y, and take a particular allocation of these output levels as between A and B – there will correspond a particular level of utility for A and for B, which can be represented as a point in U^A/U^B space, such as R in Figure 4.5. Given fixed amounts of capital and labour, not all points in U^A/U^B space are feasible. Suppose that all available resources were used to produce commodities solely for consumption by A, and that the combination of X and Y then produced was such as to maximise A's utility. Then, the corresponding point in utility space would be U^A_{max} in Figure 4.5. With all production serving the interests of B, the corresponding point would be U^B_{max}. The area bounded by U^A_{max} 0 U^B_{max} is the utility possibility set – given its resources, production technologies and preferences, the economy can deliver all combinations of U^A and U^B lying in that area. The line U^A_{max} U^B_{max} is the utility possibility frontier – the economy cannot deliver combinations of U^A and U^B lying outside that line. The shape of the utility possibility frontier depends on the particular forms of the utility and production functions, so the way in which it is represented in Figure 4.5 is merely one possibility. However, for the usual assumptions about utility and production functions, it would be generally bowed outwards in the manner shown in Figure 4.5.

The utility possibility frontier is the locus of all possible combinations of U^A and U^B that correspond to efficiency in allocation. Consider the point R in Figure 4.5, which is inside the utility possibility frontier. At such a point, there are possible reallocations that could mean higher utility for both A and B. By securing allocative efficiency, the economy could, for example, move to a point on the frontier such as Z. But, given its endowments of capital and labour, and the production and utility functions, it could not continue northeast beyond the frontier. Only U^A/U^B combinations lying along, or inside, the frontier are feasible. The move from R to Z would be a Pareto improvement. So would be move from R to T, or to S, or to any point along the frontier between T and S.

The utility possibility frontier shows the U^A/U^B combinations that correspond to efficiency in allocation, situations where there is no scope for a Pareto improvement. There are many such combinations. Is it possible, using the information available, to say which of the points on the frontier is best from the point of view of society? It is not possible, for the simple reason that the criterion of economic efficiency does not provide any basis for making interpersonal comparisons. Put another way, efficiency does not give us a criterion for judging which allocation is best from a social point of view. Choosing a point along the utility possibility frontier is about making moves that must involve making one individual worse off in order to make the other better off. Efficiency criteria do not cover such choices.

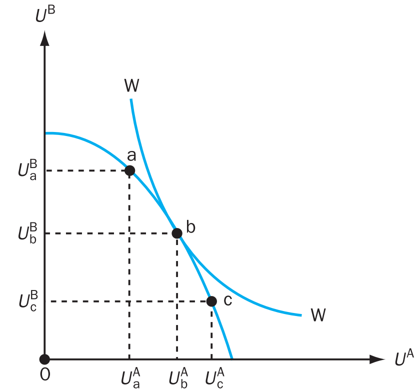


Figure 4.6 Maximised social welfare

4.3 The social welfare function and optimality

In order to consider such choices we need the concept of a social welfare function, SWF, which was introduced in the Chapter 3. A SWF can be used to rank alternative allocations. For the two-person economy that we are examining, a SWF will be of the general form:

$$W = W(U^A, U^B) \tag{4.6}$$

The only assumption that we make here regarding the form of the SWF is that welfare is non-decreasing in U^A and U^B. That is, for any given level of U^A, welfare cannot decrease if U^B were to rise and vice versa. In other words, we assume that $W_A = \partial W/\partial U^A$ and $W_B = \partial W/\partial U^B$ are both positive. Given this, the SWF is formally of the same nature as a utility function. Whereas the latter associates numbers for utility with combinations of consumption levels X and Y, a SWF associates numbers for social welfare with combinations of utility levels U^A and U^B. Just as we can depict a utility function in terms of indifference curves, so we can depict a SWF in terms of social welfare indifference curves. Figure 4.6 shows a social welfare indifference curve WW, which has the same slope as the utility possibility frontier at b, which point identifies the combination of U^A and U^B that maximises the SWF.

The reasoning which establishes that b corresponds to the maximum of social welfare that is attainable should be familiar by now – points to the left or the right of b on the utility possibility frontier, such as a and c, must be on a lower social welfare indifference curve, and points outside of the utility possibility frontier are not attainable. The fact that the optimum lies on the utility possibility frontier means that all of the necessary conditions for efficiency must hold at the optimum. Conditions 4.3, 4.4 and 4.5 must be satisfied for the maximisation of welfare. Also, an additional condition, the equality of the slopes of a social indifference curve and the utility possibility frontier, must be satisfied. This condition can be stated, as established in Appendix 4.1, as

$$\frac{W_A}{W_B} = \frac{U_X^B}{U_X^A} = \frac{U_Y^B}{U_Y^A} \tag{4.7}$$

The left-hand side here is the slope of the social welfare indifference curve. The two right-hand-side terms are alternative expressions for the slope of the utility possibility frontier. At a social welfare maximum, the slopes of the indifference curve and the frontier must be equal, so that it is not possible to increase social welfare by transferring goods, and hence utility, between persons.

While allocative efficiency is a necessary condition for optimality, it is not generally true that moving from an allocation that is not efficient to one that is efficient must represent a welfare improvement. Such a move might result in a lower level of social welfare. This possibility is illustrated in Figure 4.7. At C the allocation is not efficient, at D it is. However, the allocation at C gives a higher level of social welfare than does that at D. Having made this point, it should also be said that whenever there is an inefficient allocation, there is always some other allocation which is both efficient and superior in welfare terms. For example, compare points C and E. E is allocatively efficient while C is not, and E is on a higher

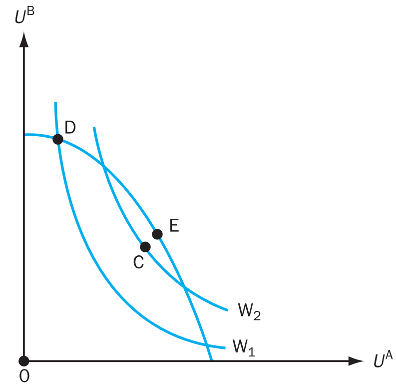


Figure 4.7 Welfare and efficiency

social welfare indifference curve. The move from C to E is a Pareto improvement where both A and B gain, and hence involves higher social welfare. On the other hand, going from C to D replaces an inefficient allocation with an efficient one, but the change is not a Pareto improvement – B gains but A suffers – and involves a reduction in social welfare. Clearly, any change which is a Pareto improvement must increase social welfare as defined here. Given that the SWF is non-decreasing in U^A and U^B, increasing U^A/U^B without reducing U^B/U^A must increase social welfare. For the kind of SWF employed here, a Pareto improvement is an unambiguously good thing (subject to the possible objections to preference-based utilitarianism noted in Chapter 3, of course). It is also clear that allocative efficiency is a good thing (subject to the same qualification) if it involves an allocation of commodities as between individuals that can be regarded as fair. Judgements about fairness, or equity, are embodied in the SWF in the analysis here. If these are acceptable, then optimality is an unambiguously good thing. In Part 2 of this chapter we look at the way markets allocate resources and commodities. To anticipate, we shall see that what can be claimed for markets is that, given ideal institutional arrangements and certain modes of behaviour, they achieve allocative efficiency. It cannot be claimed that, alone, markets, even given ideal institutional arrangements, achieve what might generally or reasonably be regarded as fair allocations. Before looking at the way markets allocate resources, we shall look at economists' attempts to devise criteria for evaluating alternative allocations that do not involve explicit reference to a social welfare function.

4.4 Compensation tests

If there were a generally agreed SWF, there would be no problem, in principle, in ranking alternative allocations. One would simply compute the value taken by the SWF for the allocations of interest, and rank by the computed values. An allocation with a higher SWF value would be ranked above one with a lower value. There is not, however, an agreed SWF. The relative weights to be assigned to the utilities of different individuals are an ethical matter. Economists prefer to avoid specifying the SWF if they can. Precisely the appeal of the Pareto improvement criterion – a reallocation is desirable if it increases somebody's utility without reducing anybody else's utility – is that it avoids the need to refer to the SWF to decide on whether or not to recommend that reallocation. However, there are two problems, at the level of principle, with this criterion. First, as we have seen, the recommendation that all reallocations satisfying this condition be undertaken does not fix a unique allocation. Second, in considering policy issues there will be very few proposed reallocations that do not involve some individuals gaining and some losing. It is only rarely, that is, that the welfare economist will be asked for advice about a reallocation that improves somebody's lot without damaging somebody else's. Most reallocations that require analysis involve winners and losers and are, therefore, outside of the terms of the Pareto improvement criterion.

Given this, welfare economists have tried to devise ways, which do not require the use of an SWF, of comparing allocations where there are winners and losers. These are compensation tests. The basic idea is simple. Suppose there are two allocations, denoted 1 and 2, to be compared. As previously, the essential ideas are covered if we consider a two-person two commodity world. Moving from allocation 1 to allocation 2 involves one individual gaining and the other losing. The Kaldor compensation test, named after its originator, says that allocation 2 is superior

Table 4.1 Two tests, two answers

	Allocation 1			Allocation 2		
	X	*Y*	*U*	*X*	*Y*	*U*
A	10	5	50	20	5	100
B	5	20	100	5	10	50

Table 4.2 Two tests, one answer

	Allocation 1			Allocation 2		
	X	*Y*	*U*	*X*	*Y*	*U*
A	10	5	50	20	10	200
B	5	20	100	5	10	50

to allocation 1 if the winner could compensate the loser and still be better off. Table 4.1 provides a numerical illustration of a situation where the Kaldor test has 2 superior to 1. In this, constructed, example, both individuals have utility functions that are $U = XY$, and A is the winner for a move from 1 to 2, while B loses from such a move. According to the Kaldor test, 2 is superior because at 2 A could restore B to the level of utility that she enjoyed at 1 and still be better off than at 1. Starting from allocation 2, suppose that 5 units of X were shifted from A to B. This would increase B's utility to 100 (10×10), and reduce A's utility to 75 (15×5) – B would be as well off as at 1 and A would still be better off than at 1. Hence, the argument is, allocation 2 must be superior to 1, as if such a reallocation were undertaken the benefits as assessed by the winner would exceed the losses as assessed by the loser. Note carefully that this test does not require that the winner actually does compensate the loser. It requires only that the winner could compensate the loser, and still be better off. For this reason, the Kaldor test, and the others to be discussed below, are sometimes referred to as 'potential compensation tests'. If the loser was actually fully compensated by the winner, and the winner was still better off, then we would be looking at a situation where there was a Pareto improvement.

The numbers in Table 4.1 have been constructed so as to illustrate a problem with the Kaldor test. This is that it may sanction a move from one allocation to another, but that it may also sanction a move from the new allocation back to the original allocation. Put another way, the problem is that if we use the Kaldor test to ask whether 2 is superior to 1 we may get a 'yes', and we may also get a 'yes' if we ask if 1 is superior to 2. Starting from 2 and considering a move to 1, B is the winner and A is the loser. Looking at 1 in this way, we see that if 5 units of Y were transferred from B to A, B would have U equal to 75, higher than in 2, and A would have U equal to 100, the same as in 2. So, according to the Kaldor test done this way, 1 is superior to 2.

This problem with the Kaldor test was noted by Hicks, who actually put things in a slightly different way. He proposed a different (potential) compensation test for considering whether the move from 1 to 2 could be sanctioned. The question in the Hicks test is: could the loser compensate the winner for forgoing the move and be no worse off than if the move took place? If the answer is 'yes', the reallocation is not sanctioned, otherwise it is on this test. In Table 4.1, suppose at allocation 1 that 5 units of Y are transferred from B, the loser from a move to 2, to A. A's utility would then go up to 100 (10×10), the same as in allocation 2, while B's would go down to 75 (5×15), higher than in allocation 2. The loser in a reallocation from 1 to 2, could, that is, compensate the individual who would benefit from such a move for it not actually taking place, and still be better off than if the move had taken place. On this test, allocation 1 is superior to allocation 2.

In the example of Table 4.1, the Kaldor and Hicks (potential) compensation tests give different answers about the rankings of the two allocations under consideration. This will not be the case for all reallocations that might be considered. Table 4.2 is a, constructed, example where both tests give the same answer. For the Kaldor test, looking at 2, the winner A could give the loser B 5 units of X and still be better off than at 1 ($U = 150$), while B would then be fully compensated for the loss involved in going from 1 to 2 ($U = 10 \times 10 = 100$). On this test, 2 is superior to 1. For the Hicks test, looking at 1, the most that the loser B could transfer to the winner A so as not to be worse off than in allocation 2 is 10 units of Y. But, with 10 of X and 15 of Y, A would have $U = 150$, which is less than A's utility at 2, 200. The loser could not compensate the winner for forgoing the move and be no worse off than if the move took place, so again 2 is superior to 1.

For an unambiguous result from the (potential) compensation test, it is necessary to use both the Kaldor and the Hicks criteria. The Kaldor–Hicks–Scitovsky test – known as such because Scitovsky pointed out that both criteria are required – says that a reallocation is desirable if:

(i) the winners could compensate the losers and still be better off

and

(ii) the losers could not compensate the winners for the reallocation not occurring and still be as well off as they would have been if it did occur.

In the example of Table 4.2 the move from 1 to 2 passes this test, in that of Table 4.1 it does not.

As we shall see, especially in Chapters 11 and 12 on cost–benefit analysis and environmental valuation respectively, compensation tests inform much of the application of welfare economics to environmental problems. Given that utility functions are not observable, the practical use of compensation tests does not take the form worked through here, of course. Rather, as we shall see, welfare economists work with monetary measures which are intended to measure utility changes. As noted above, the attraction of compensation tests is that they do not require reference to a SWF. However, while they do not require reference to a SWF, it is not the case that they solve the problem that the use of a SWF addresses. Rather, compensation tests simply ignore the problem. As indicated in the examples above, compensation tests treat winners and losers equally. No account is taken of the fairness of the distribution of well-being.

Consider the example in Table 4.3. Considering a move from 1 to 2, A is the loser and B is the winner. As regards (i), at 2 moving one unit of Y from B to A would make A as well off as she was at 1, and would leave B better off (U = 225) than at 1. As regards (ii), at 1 moving either two of X or one of Y from A to B would leave A as well off as at 2, but in neither case would this be sufficient to compensate B for being at 1 rather than 2 (for B after such transfers U = 140 or U = 105). According to both (i) and (ii) 2 is superior to 1, and such a reallocation passes the Kaldor–Hicks–Scitovsky test. Note, however, that A is the poorer of the two individuals, and that the reallocation sanctioned by the compensation test makes A worse off, and makes B better off. In sanctioning such a reallocation, the compensation test is saying either that fairness is irrelevant or that there is an implicit SWF such that the reallocation is consistent with the notion of fairness that it embodies. If, for example, the SWF was

$$W = 0.5U^{A} + 0.5U^{B}$$

then at 1 welfare would be 75 and at 2 it would be 140. Weighting A's losses equally with B's gains means that 2 is superior to 1 in welfare terms. If it were thought appropriate to weight A's losses much more heavily than B's gains, given that A is relatively poor, then using, say

$$W = 0.95U^{A} + 0.05U^{B}$$

gives welfare at 1 as 52.5 and at 2 as 50, so that 1 is superior to 2 in welfare terms, notwithstanding that the move from 1 to 2 is sanctioned by the (potential) compensation test.

In the practical use of compensation tests in applied welfare economics, welfare, or distributional, issues are usually ignored. The monetary measures of winners' gains (benefits) and losers' losses (costs) are usually given equal weights irrespective of the income and wealth levels of those to whom they accrue. In part, this is because it is often difficult to identify winners and losers sufficiently closely to be able to say what their relative income and wealth levels are. But, even in those cases where it is clear that, say, costs fall mainly on the relatively poor and benefits mainly on the better off, economists are reluctant to apply welfare weights when applying a compensation test by comparing total gains and total losses – they simply report on whether or not £s of gain exceed £s of loss. Various justifications are offered for this practice. First, at the level of principle, that there is no generally agreed SWF for them to use, and it would be inappropriate for economists to

Table 4.3 Compensation may not produce fairness

	Allocation 1			Allocation 2		
	X	Y	U	X	Y	U
A	10	5	50	10	4	40
B	5	20	100	15	16	240

themselves specify a SWF. Second, that, as a practical matter, it aids clear thinking to separate matters of efficiency from matters of equity, with the question of the relative sizes of gains and losses being treated as an efficiency issue, while the question of their incidence across poor and rich is an equity issue. On this view, when considering some policy intended to effect a reallocation the job of the economic analyst is to ascertain whether the gains exceed the losses. If they do, the policy can be recommended on efficiency grounds, and it is known that the beneficiaries could compensate the losers. It is a separate matter, for government, to decide whether compensation should actually occur, and to arrange for it to occur if it is thought desirable. These matters are usually considered in the context of a market economy, and we shall return to them in that context at the end of the next section of the chapter.

PART II ALLOCATION IN A MARKET ECONOMY

4.5 Efficiency given ideal conditions

A variety of institutional arrangements might be employed to allocate resources, such as dictatorship, central planning or free markets. Any of these could, in principle, achieve an efficient allocation of resources. Here, we are particularly interested in the consequences of free market resource allocation decisions. This is for three, related, reasons. First, for dictatorship and central planning to achieve allocative efficiency it is necessary that the dictator/central planner knows all of the economy's production and utility functions. This is clearly infeasible, and is one of the reasons that attempts to run economies in these ways have been unsuccessful. The great attraction of free markets as a way of organising economic activity is that they do not require that any institution or agent has such knowledge. That is the second reason for our concentration on markets – they are decentralised information-processing systems of great power. The third reason is that the modern welfare economics that is the basis for environmental and resource economics takes it that markets are the way economies are mainly organised. Environmental and resource issues are studied, that is, as they arise in an economy where markets are the dominant social institution for organising production and consumption. The market economy is now the dominant mode of organising production and consumption in human societies.

Welfare economics theory points to a set of circumstances such that a system of free markets would sustain an efficient allocation of resources. These 'institutional arrangements', as we shall call them, include the following:

1. Markets exist for all goods and services produced and consumed.
2. All markets are perfectly competitive.
3. All transactors have perfect information.
4. Private property rights are fully assigned in all resources and commodities.
5. No externalities exist.
6. All goods and services are private goods. That is, there are no public goods.
7. All utility and production functions are 'well behaved'.[3]

In addition to these institutional arrangements, it is necessary to assume that the actors in such a system – firms and individuals, often referred to jointly as 'economic agents' or just 'agents' – behave in certain ways. It is assumed that agents always strive to do the best for themselves that they can in the circumstances that they find themselves. Firms are assumed to maximise profits, individuals to maximise utility. A shorthand way of saying this is to say that all agents are maximisers.

An efficient allocation would be the outcome in a market economy populated entirely by maximisers and where all of these institutional arrangements were in place. Before explaining why and how this

[3] For a full account of what 'well behaved' means the reader is referred to one the welfare economics texts cited in the Further Reading section at the end of the chapter. Roughly, in regard to utility it means that indifference curves are continuous and have the 'bowed toward the origin' shape that they are usually drawn with in the textbooks. In regard to production, the main point is that increasing returns to scale are ruled out.

is so, a few brief comments on these conditions required for a market system to be capable of realising allocative efficiency are in order. First, note that, as we shall see in later sections of this chapter where we discuss public goods and externalities, 5 and 6 are really particulars of 4. Second, note that 4 is necessary for 1 – markets can only work where there are private property rights and a justice system to enforce and protect such rights. Third, that an important implication of 2 is that buyers and sellers act as 'price-takers', believing that the prices that they face cannot be influenced by their own behaviour. No agent, that is, acts in the belief that he or she has any power in the market. Finally, note that these are a very stringent set of conditions, which do not accurately describe any actual market economy. The economy that they do describe is an ideal type, to be used in the welfare analysis of actual economies as a benchmark against which to assess performance, and to be used to devise policies to improve the performance, in regard to efficiency criteria, of such actual economies.

We now explain why a market allocation of resources would be an efficient allocation in such ideal circumstances. A more formal treatment is provided in Appendix 4.2.

Consider first individuals and their consumption of produced commodities. Any one individual seeks to maximise utility given income and the, fixed, prices of commodities. Figure 4.8, familiar from introductory microeconomics, refers to an individual in a two-commodity economy. The line $Y_{max}X_{max}$ is the budget constraint. Y_{max} is the amount of Y available if all income is spent on Y, X_{max} is consumption if all income is spent on X. The slope of the budget constraint gives the price ratio P_X/P_Y. Utility maximisation requires consumption X^* and Y^* corresponding to point b on the indifference curve U^*U^*. Consumption at points on $Y_{max}X_{max}$ to the left or right of b, such as a and c, would mean being on a lower indifference curve than U^*U^*. Consumption patterns corresponding to points to the northeast of $Y_{max}X_{max}$ are not attainable with the given income and prices. The essential characteristic of b is that the budget line is tangential to an indifference curve. This means that the slope of the indifference curve is equal to the price ratio. Given that the slope of the indifference curve is the MRUS, we have:

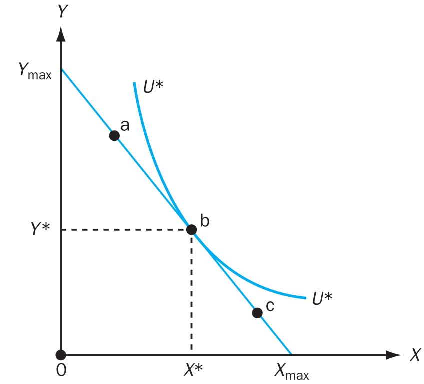


Figure 4.8 Utility maximisation

$$\text{MRUS} = \frac{P_X}{P_Y}$$

In the ideal conditions under consideration, all individuals face the same prices. So, for the two-individual, two-commodity market economy, we have

$$\text{MRUS}^{\text{A}} = \text{MRUS}^{\text{B}} = \frac{P_X}{P_Y} \tag{4.8}$$

Comparison of equation 4.8 with equation 4.3 shows that the consumption efficiency condition is satisfied in this ideal market system. Clearly, the argument here generalises to many-person, multi-commodity contexts.

Now consider firms. To begin, instead of assuming that they maximise profits, we will assume that they minimise the costs of producing a given level of output. The cost-minimisation assumption is in no way in conflict with the assumption of profit maximisation. On the contrary, it is implied by the profit-maximisation assumption, as, clearly, a firm could not be maximising its profits if it was producing whatever level of output that involved at anything other than the lowest possible cost. We are leaving aside, for the moment, the question of the determination of the profit-maximising level of output, and focusing instead on the prior question of cost minimisation for a given level of output. This question is examined in Figure 4.9, where X^*X^* is the isoquant corresponding to some given output level X^*. The straight lines K_1L_1, K_2L_2, and K_3L_3 are isocost lines. For given prices for inputs, P_K and P_L, an isocost line shows the combinations of input levels for

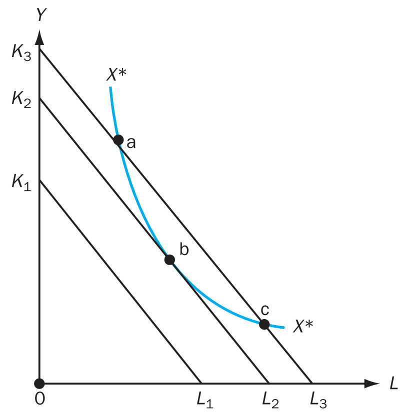


Figure 4.9 Cost minimisation

K and L that can be purchased for a given total expenditure on inputs. K_3L_3 represents, for example, a higher level of expenditure on inputs, greater cost, than K_2L_2. The slope of an isocost line is the ratio of input prices, P_K/P_L. Given production of X^*, the cost-minimising firm will choose the input combination given by the point b. Any other combination, such as a or c, lying along X^*X^* would mean higher total costs. Combinations represented by points lying inside K_2L_2 would not permit of the production of X^*. The essential characteristic of b is that an isocost line is tangential to, has the same slope as, an isoquant. The slope of an isoquant is the MRTS so that cost-minimising choices of input levels must be characterised by:

$$\text{MRTS} = \frac{P_K}{P_L}$$

In the ideal circumstances under consideration, all firms, in all lines of production, face the same P_K and P_L, which means that

$$\text{MRTS}_X = \text{MRTS}_Y \tag{4.9}$$

which is the same as equation 4.4, the production efficiency condition for allocative efficiency – cost-minimising firms satisfy this condition.

The remaining condition that needs to be satisfied for allocative efficiency to exist is the product-mix condition, equation 4.5, which involves both individuals and firms. In explaining how this condition is satisfied in an ideal market system we will also see how the profit-maximising levels of production are determined. Rather than look directly at the profit-maximising output choice, we look at the choice of input levels that gives maximum profit. Once the input levels are chosen, the output level follows from the production function. Consider the input of labour to the production of X, with marginal product X_L. Choosing the level of X_L to maximise profit involves balancing the gain from using an extra unit of labour against the cost of so doing. The gain here is just the marginal product of labour multiplied by the price of output, i.e. P_XX_L. The cost is the price of labour, i.e. P_L. If P_L is greater than P_XX_L, increasing labour use will reduce profit. If P_L is less than P_XX_L, increasing labour use will increase profit. Clearly, profit is maximised where $P_L = P_XX_L$.

The same argument applies to the capital input, and holds in both lines of production. Hence, profit maximisation will be characterised by

$$P_XX_L = P_L$$

$$P_XX_K = P_K$$

$$P_YY_L = P_L$$

$$P_YY_K = P_K$$

which imply

$$P_XX_L = P_YY_L = P_L$$

and

$$P_XX_K = P_YY_K = P_K.$$

Using the left-hand equalities here, and rearranging, this is

$$\frac{P_X}{P_Y} = \frac{Y_L}{X_L} \tag{4.10.a}$$

and

$$\frac{P_X}{P_Y} = \frac{Y_K}{X_K} \tag{4.10.b}$$

Now, the right-hand sides here are MRT_L and MRT_K, as they are the ratios of marginal products in the two lines of production and hence give the terms on which the outputs change as labour and capital are shifted between industries. Given that the left-hand sides in equations 4.10.a and 4.10.b are the same, we can write

$$\text{MRT}_L = \text{MRT}_K = \frac{P_X}{P_Y} \tag{4.11}$$

showing that the marginal rate of transformation is the same for labour shifting as for capital shifting. Referring back to equation 4.8, we can now write

$$\text{MRT}_L = \text{MRT}_K = \frac{P_X}{P_Y} = \text{MRUS}^{\text{A}} = \text{MRUS}^{\text{B}} \quad (4.12)$$

showing that the profit-maximising output levels in the ideal market economy satisfy the product-mix condition for allocative efficiency, equation 4.5.

This completes the demonstration that in an ideal market system the conditions necessary for allocative efficiency will be satisfied. We conclude this section by looking briefly at profit-maximising behaviour from a perspective that will be familiar from an introductory microeconomics course. There, students learn that in order to maximise profit, a firm which is a price-taker will expand output up to the level at which price equals marginal cost. Figure 4.10 refers. For output levels below X^*, price exceeds marginal cost so that increasing output will add more to receipts than to costs, so increasing profit as the difference between receipts and costs. For output levels greater than X^*, marginal cost exceeds price, and reducing output would increase profit. This is in no way inconsistent with the discussion above of choosing input levels so as to maximise profit. It is just a different way of telling the same story. In order to increase output, assuming technical efficiency, more of at least one input must be used. In thinking about whether or not to increase output the firm considers increasing the input of capital or labour, in the manner described above. For the case of labour in the production of X, for example, the profit-maximising condition was seen to be $P_L = P_X X_L$, which can be written as

$$\frac{P_L}{X_L} = P_X$$

which is just marginal cost equals price, because the left-hand side is the price of an additional unit of labour divided by the amount of output produced by that additional unit. Thus if the wage rate is £5 per hour, and one hour's extra labour produces 1 tonne of output, the left-hand side here is £5 per tonne, so the marginal cost of expanding output by one tonne is £5. If the price that one tonne sells for is greater/less than £5, it will pay in terms of profit to increase/decrease output by one tonne by increasing the use of labour. If the equality holds and the output price is £5, profit is being maximised. The same argument goes through in the case of capital, and the marginal cost equals price condition for profit maximisation can also be written as

$$\frac{P_K}{X_K} = P_X$$

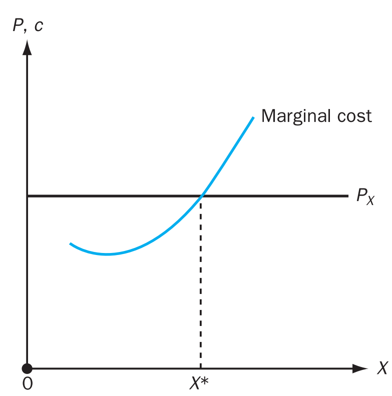


Figure 4.10 Profit maximisation

4.6 Partial equilibrium analysis of market efficiency

In examining the concepts of efficiency and optimality, we have used a general equilibrium approach. This looks at all sectors of the economy simultaneously. Even if we were only interested in one part of the economy – such as the production and consumption of cola drinks – the general equilibrium approach requires that we look at all sectors. In finding the allocatively efficient quantity of cola, for example, the solution we get from this kind of exercise would give us the efficient quantities of all goods, not just cola.

There are several very attractive properties of proceeding in this way. Perhaps the most important of these is the theoretical rigour it imposes. In developing economic theory, it is often best to use general equilibrium analysis. Much (although by no means all) of the huge body of theory that makes up resource and environmental economics analysis has such a general approach at its foundation.

But there are penalties to pay for this rigour. Doing applied work in this way can be expensive and time consuming. And in some cases data limitations make it impossible. The exercise may not be quite as daunting as it sounds, however. We could define categories in such a way that there are just two goods in the economy: cola and a composite good that is everything except cola. Indeed, this kind of 'trick' is commonly used in economic analysis. But even with this type of simplification, a general equilibrium approach is likely to be difficult and costly, and may be out of all proportion to the demands of some problem for which we seek an approximate solution.

Given the cost and difficulty of using this approach for many practical purposes, many applications use a different framework that is much easier to operationalise. This involves looking at only the part of the economy of direct relevance to the problem being studied. Let us return to the cola example, in which our interest lies in trying to estimate the efficient amount of cola to be produced. The partial approach examines the production and consumption of cola, ignoring the rest of the economy. It begins by identifying the benefits and costs to society of using resources to make cola. Then, defining *net* benefit as total benefit minus total cost, an efficient output level of cola would be one that maximises net benefit.

Let X be the level of cola produced and consumed, measured along the horizontal axis in Figure 4.11, where the vertical axis measures units of money, £s say. Panel (a) of Figure 4.11 shows the total benefits of cola (labelled B) and the total costs of cola (labelled C) for various possible levels of cola production. The reason we have labelled the curves $B(X)$ and $C(X)$, not just B and C, is to make it clear that benefits and costs each depend on, are functions of, X. Benefits and costs are measured in money units. The shapes and relative positions of the curves we have drawn for B and C are, of course, just stylised representations of what we expect them to look like. A researcher trying to answer the question we posed above would have to estimate the shapes and positions of these functions from whatever evidence is available, and they may differ from those drawn in the diagram. However, the reasoning that follows is not conditional on the particular shapes

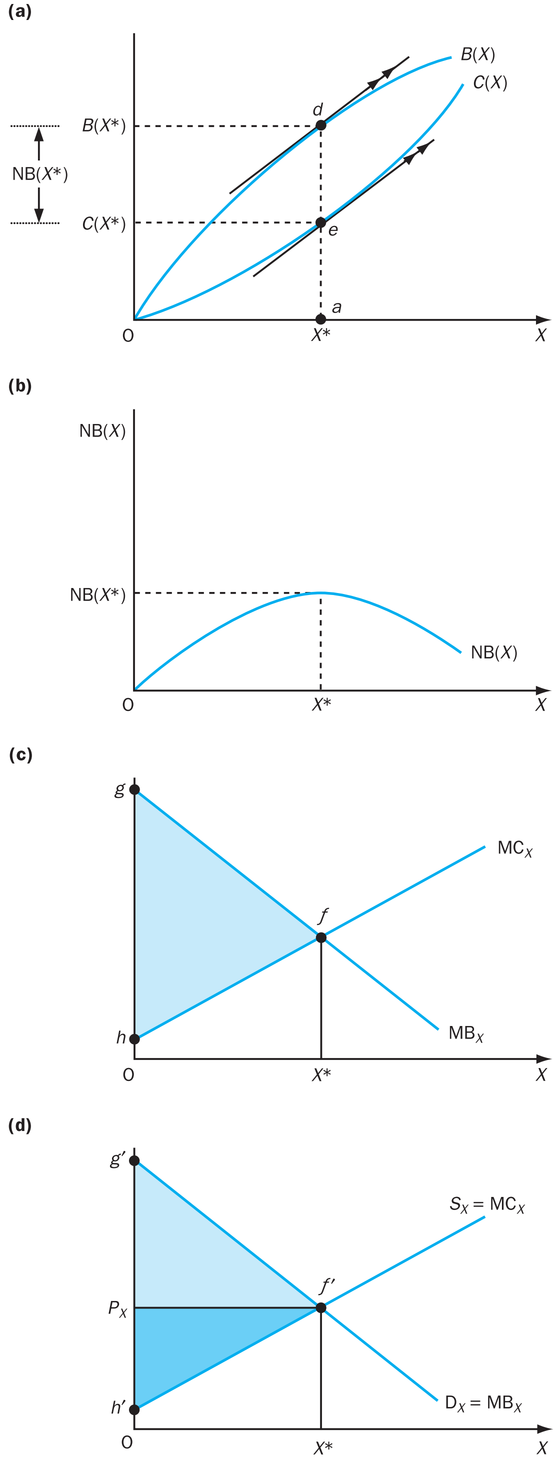


Figure 4.11 A partial equilibrium interpretation of economic efficiency

and positions that we have used, which are chosen mainly to make the exposition straightforward.

Given that we call an outcome that maximises net benefits efficient, it is clear from Figure 4.11(a) that X^* is the efficient level of cola production. Net benefits (indicated by the distance *de*) are at their maximum at that level of output. This is also shown in Figure 4.11(b), which plots the *net* benefits for various levels of X. Observe the following points:

- At the efficient output level X^* the total benefit and total cost curves are parallel to one another (Figure 4.11(a)).
- The net benefit function is horizontal at the efficient output level (Figure 4.11(b)).

The distance *de*, or equivalently the magnitude NB(X^*), can be interpreted in efficiency terms. It is a measure, in money units, of the efficiency gain that would come about from producing X^* cola compared with a situation in which no cola was made.

These ideas are often expressed in a different, but exactly equivalent way, using marginal rather than total functions. As much of the environmental economics literature uses this way of presenting ideas (and we shall do so also in several parts of this book), let us see how it is done. We use MC_X to denote the marginal cost of X, and MB_X denotes the marginal benefit of X. In Figure 4.11(c), we have drawn the marginal functions which correspond to the total functions in Figure 4.11(a). We drew the curves for $B(X)$ and $C(X)$ in Figure 4.11(a) so that the corresponding marginal functions are straight lines, a practice that is often adopted in partial equilibrium treatments of welfare economics. This is convenient and simplifies exposition of the subsequent analysis. But, the conclusions do not depend on the marginal functions being straight lines. The results to be stated hold so long as marginal benefits are positive and declining with X and marginal costs are positive and increasing with X – as they are in Figure 4.11(c).

In Figure 4.11(c) we show X^*, the cola output level that maximises net benefit, as being the level of X at which MC_X is equal to MB_X. Why is this so? Consider some level of X below X^*. This would involve MB_X greater than MC_X, from which it follows that increasing X would increase benefit by more than cost. Now consider some level of X greater than X^*, with MC_X greater than MB_X, from which it follows that reducing X would reduce cost by more than benefit, i.e. increase net benefit. Clearly, considering X levels above or below X^* in this way, it is X^* that maximises net benefit.

Can we obtain a measure of maximised net benefits from Figure 4.11(c) that corresponds to the distance *de* in Figure 4.11(a)? Such a measure is available; it is the area of the triangle *gfh*. The area beneath a marginal function over some range gives the value of the change in the total function for a move over that range. So the area beneath MB_X over the range $X = 0$ to $X = X^*$ gives us the total benefits of X^* cola (i.e. B^*), which is equal to the distance *ad* in Figure 4.11(a). Similarly, the area beneath MC_X over the range $X = 0$ to $X = X^*$ gives us the total cost of X^* (i.e. C^*), which is the same as the distance *ae* in Figure 4.11(a). By subtraction we find that the area *gfh* in Figure 4.11(c) is equal to the distance *de* in Figure 4.11(a).

Now we turn to the partial equilibrium version of the demonstration that an ideal market system maximises net benefit and secures allocative efficiency. We assume that all of the institutional arrangements listed in the previous section apply, and that all agents are maximisers. Then all those who wish to drink cola will obtain it from the market, and pay the going market price. The market demand curve, D_X, for cola will be identical to the MB_X curve, as that describes consumers' *willingness to pay* for additional units of the good – and that is exactly what we mean by a demand curve. Under our assumptions, cola is produced by a large number of price-taking firms in a competitive market. The market supply curve, S_X, is identical to the curve MC_X in Figure 4.11(c) because, given that firms produce where price equals marginal cost, the supply curve is just the marginal cost curve – each point on the supply curve is a point where price equals marginal cost. S_X shows the cost of producing additional (or marginal) cans of cola at various output levels.

The market demand and supply curves are drawn in Figure 4.11(d). When all mutually beneficial transactions have taken price, the equilibrium market price of the good will be P_X, equal at the margin to both

- consumers' subjective valuations of additional units of the good (expressed in money terms); and
- the costs of producing an additional unit of the good.

Put another way, all consumers face a common market price P_X, and each will adjust her consumption until her marginal utility (in money units) is equal to that price. Each firm faces that same fixed market price, and adjusts its output so that its marginal cost of production equals that price. So we have:

$$P_X = \text{MC}_X = \text{MB}_X \tag{4.13}$$

The equality at the margin of costs and benefits shows that cola is being produced in the amount consistent with the requirements of allocative efficiency. We must emphasise here something that it is sometimes possible to forget when using partial equilibrium analysis. The fact that equation 4.13 holds for the cola, or whatever, market means that the quantity of cola, or whatever, produced and consumed is consistent with allocative efficiency only if *all* the institutional arrangements listed at the start of this section are in place. It is necessary, for example, not only that the cola market be perfectly competitive, but also that all markets be perfectly competitive. And, it is necessary, for example, that all inputs to and outputs from production be traded in such markets. If such requirements are not met elsewhere in the economy, the supply and demand curves in the cola market will not properly reflect the costs and benefits associated with different levels of cola production. Some of the issues arising from these remarks will be dealt with in the last part of this chapter, under the heading of 'market failure'.

Finally here, we can use Figure 4.11(d) to introduce the concepts of *consumers' surplus* and *producers' surplus*, which are widely used in welfare economics and its application to environmental and natural resource issues. The area beneath the demand curve between zero and X^* units of the good shows the total consumers' willingness to pay, WTP, for X^* cans of cola per period. To see this, imagine a situation in which cans of cola are auctioned, one at a time. The price that the first can offered would fetch is given by the intercept of the demand curve, $0g'$. As successive cans are offered so the price that they fetch falls, as shown by the demand curve. If we add up all the prices paid until we get to X^*, and recognising that X^* is a very large number of cans, we see that the total revenue raised by the auction process which stops at X^* will be the area under the demand curve over $0X^*$, i.e. $0g'f'X^*$. But this is not the way the market works. Instead of each can being auctioned, a price is set and all cans of cola demanded are sold at that price. So, the individual who would have been willing to pay $0g'$ for a can actually gets it for P_X. Similarly, the individual who would have been willing to pay just a little less than $0g'$ actually pays P_X. And so and so on, until we get to the individual whose WTP is P_X, and who also actually pays P_X. All individuals whose WTP is greater than P_X are, when all cans sell at P_X, getting a surplus which is the excess of their WTP over P_X. Consumers' surplus is the total of these individual surpluses, the area between the demand curve and the price line over $0X^*$, i.e. $P_Xg'f'$. Another way of putting this is that consumers' surplus is the difference between total willingness to pay and total actual expenditure, which is the difference between area $0g'f'X^*$ and area $0P_Xf'X^*$, which is the area of the triangle $P_Xg'f'$.

Producers' surplus in Figure 4.11(d) is the area of the triangle $h'P_Xf'$. The reasoning to this is very similar to that for consumers' surplus. As noted above, the supply curve is, given the ideal conditions being assumed here, just the marginal cost curve. The first can of cola costs $0h'$ to produce, but sells in the market for P_X, so there is a surplus of $h'P_X$. The surplus on the production of each further can is given by the vertical distance from the price line to the supply curve. The sum of all these vertical distances is total producers' surplus, the area $h'P_Xf'$. An alternative way of putting this is that total revenue is the area $0P_Xf'X^*$, while total cost is $0h'f'X^*$, so that producers' surplus is revenue minus costs, i.e. $h'P_Xf'X^*$.

4.7 Market allocations are not necessarily equitable

The previous sections have shown that, provided certain conditions are satisfied, a system of free markets will produce an allocation that is efficient in the sense that nobody can be made better off except at the cost of making at least one other person worse off. It has not been shown that a system of free markets will produce an optimal allocation according to any particular social welfare function.

The basic intuition of both the positive – the attainment of efficiency – and the negative – no necessary attainment of equity – here is really rather simple. The essential characteristic of markets is voluntary exchange. Think of two individuals who meet, each carrying a box containing an assortment of commodities. The two assortments are different. The two individuals lay out the contents of their boxes, and swap items until there are no further swaps that both see as advantageous. Then, considering just these two individuals and the collection of commodities jointly involved, the allocation of that collection at the end of the swapping is efficient in the sense that if somebody else came along and forced them to make a further swap, one individual would feel better off but the other worse off, whereas prior to the enforced swap both felt better off than they did with their initial bundles. The attainment of efficiency is simply the exhaustion of the possibilities for mutually beneficial exchange. Clearly, if one individual's box had been several times as large as the other's, if one individual had a much larger initial endowment, we would not expect the voluntary trade process to lead to equal endowments. Voluntary trade on the basis of self-interest is not going to equalise wealth. Further, it is also clear that as the initial endowments of the two individuals – the sizes of their boxes and their contents – vary, so will the positions reached when all voluntary swaps have been made.

The formal foundations for modern welfare economics and its application to policy analysis in market economies are two fundamental theorems. These theorems take it that all agents are maximisers, and that the ideal institutional conditions stated at the start of this section hold. The first states that a competitive market equilibrium is an efficient allocation. Basically, this is saying that equilibrium is when there are no more voluntary exchanges, and that when there are no more voluntary exchanges all the gains from trade have been exhausted, so the situation must be one of efficiency – one where nobody can be made better off save at the cost of making somebody else worse off. The second theorem states that to every efficient allocation there corresponds a competitive market equilibrium based on a particular distribution of initial endowments. An alternative statement of this theorem, of particular relevance to policy analysis, is that any efficient allocation can be realised as a competitive market equilibrium given the appropriate set of lump-sum taxes on and transfers to individual agents. The point of the second theorem is that the efficient allocation realised by a competitive equilibrium is conditioned on the distribution of initial endowments, and that if those initial endowments are such that the resulting efficient allocation is considered inequitable, altering them by lump-sum taxes and transfers will produce another efficient allocation. If the taxes/transfers redistribute from the better to the worse off, the new efficient allocation will be more equitable.

The implication of these two theorems, which has enormous influence on the way that economists approach policy analysis in an economy mainly run by markets, is that there are two essentially separable dimensions to the economic problem. These are the problems of efficiency and equity. The theorems are taken to mean that, in effect, society can, via government, take a view on equity and achieve what it wants there by a system of redistributive taxes and payments, and then leave it to markets to achieve efficiency in allocation given the post-tax/transfer distribution of endowments. This can be put the other way round. The theorems are taken to mean that the government should not intervene in markets directly to pursue any equity objectives. It should not, for example, subsidise a commodity that figures largely in the consumption of the poor. To do so would prevent the market system attaining an efficient allocation. Anyway, it is unnecessary. The interests of the poor are to be looked after by redistributive taxes and transfers.

These theorems hold only in the ideal conditions being assumed in this part of the chapter. It will already have occurred to the reader that these conditions are not fully satisfied in any actual economy – we consider some violations and their policy implications in the next part of the chapter. It is also required that the government's redistribution is in the form of lump-sum taxes and transfers. By 'lump sum' is meant taxes and transfers that do not directly affect the incentives facing agents – in the case of taxes, for example, liability must not depend on behaviour, so that income taxes are not lump-sum taxes. Lump-sum taxes and transfers are not, in fact, widely used by governments as they are generally seen as politically infeasible.

Notwithstanding that the conditions under which the two theorems hold are not fully satisfied in any actual economy, the overwhelming majority of economists do approach practical policy analysis on the basis that the problems of efficiency and equity can be dealt with independently.

PART III MARKET FAILURE, PUBLIC POLICY AND THE ENVIRONMENT

In Part I of this chapter, we laid out the conditions that characterise an efficient allocation. In Part II, we showed that, given 'ideal' circumstances concerning institutions and behaviour, a system of markets would produce an efficient allocation. We noted that the ideal circumstances are truly ideal, in that they do not describe any actual economy. Actual market economies depart from the ideal circumstances in a variety of ways, and the allocations that they produce are not efficient. Economists use welfare economics to identify 'market failures' – situations where actual circumstances depart from the ideal – and to recommend policies to correct them so that actual economies perform better in relation to the objective of efficiency. Much of environmental and resource economics is welfare economics of this sort. It is concerned with identifying and correcting market failure in relation to the services that the environment provides to the economy. In this part of the chapter, we introduce some of the basic ideas involved here. In Part II of the book, we cover the application of the basic ideas to the problem of environmental pollution. Part III extends the basic ideas to cover intertemporal allocation problems, and then looks, mainly, at the welfare economics of the amenity services that the environment provides. Part IV of the book then deals, mainly, with the economics of natural resources as inputs to production.

4.8 The existence of markets for environmental services

To recapitulate, we have seen that for markets to produce efficient allocations, it is necessary that:

1. Markets exist for all goods and services produced and consumed.
2. All markets are perfectly competitive.
3. All transactors have perfect information.
4. Private property rights are fully assigned in all resources and commodities.
5. No externalities exist.
6. All goods and services are private goods. That is, there are no public goods.
7. All utility and production functions are 'well behaved'.
8. All agents are maximisers.

Clearly, 1 here is fundamental. If there are goods and services for which markets do not exist, then the market system cannot produce an efficient allocation, as that concept applies to all goods and services that are of interest to any agent, either as utility or production function arguments. Further, 4 is necessary for 1 – a market in a resource or commodity can only exist where there are private property rights in that resource or commodity.

We can define a property right as: a bundle of characteristics that convey certain powers to the owner of the right.[4] These characteristics concern conditions of appropriability of returns, the ability to divide or transfer the right, the degree of exclusiveness of the right, and the duration and enforceability of the right. Where a right is exclusive to one person or corporation, a private property right is said to exist.

In Chapter 2 we provided a classification of the services that the natural environment provides to economic activity, using Figure 2.1. Let us now

[4] This definition is taken from Hartwick and Olewiler (1986).

briefly consider the different classes of service distinguished there in relation to the question of the existence of private property rights. Where these do not exist, market forces cannot allocate efficiently. If efficiency is the objective, some kind of public policy intervention is required. Our remarks here are intended only to provide a general overview, as a guide to what follows in the rest of this book. The details of any particular case can be quite complicated.

In regard to the provision of inputs to production, natural resources, we made two major distinctions – between flow and stock resources, and, for the latter, between renewables and non-renewables. Generally, there are no private property rights in flow resources as such. Individuals or corporations do not, for example, have property rights in flows of solar radiation. They may, however, have property rights in land, and, hence, in the ability to capture the solar radiation falling on that land.[5] Deposits of non-renewable natural resources are, generally, subject to private property rights. Often these reside ultimately with the government, but are sold or leased by it to individuals and/or corporations. The problems arising from the non-existence of private property rights are not central to the economics of non-renewable resources.

They do, on the other hand, feature large in the renewable resource economics literature. Many, but not all, of the biotic populations exploited by man as hunter–gatherer, rather than agriculturist, are not subject to private property rights. The standard example of the case where they are not is the ocean fishery. Where private property rights are absent, two sorts of situation may obtain. In the case of 'open-access resources' exploitation is uncontrolled. The term 'common-property resources' is used whenever some legal or customary conventions, other than private property rights, regulate exploitation of the resource. Whereas an open-access regime definitely will not promote exploitation that corresponds to efficiency, a common-property regime may given the appropriate conventions and regulation. Much of the modern fisheries economics literature, as will be seen in Chapter 17, is concerned with the design of systems of government regulation of common property that will promote behaviour consistent with efficiency on the part of the private agents actually exploiting the fishery.

The second class of environmental service that was distinguished was that of receptacle for the wastes arising in economic activity. Generally, for most of history and for many wastes, the environment as waste sink has not been subject to private property rights, and has been, in effect, an open-access resource. With increasing awareness of the problems of pollution arising, states have moved to legislate so as to convert many waste sinks from open-access resources to common-property resources. Much of Part II of the book is about the economic analysis that is relevant to the public policy questions arising. What is the level of pollution that goes with efficiency? How should the behaviour of waste dischargers be regulated? We shall introduce the basic ideas involved here later in this chapter, when discussing 'externalities'.

The case of the amenity services that the environment provides is rather like that of flow resources, in that the service itself will not generally be subject to private property rights, though the means of accessing it may be. Thus, for example, nobody can own a beautiful view, but the land that it is necessary to visit in order to see it may be privately owned. Private property rights in a wilderness area would allow the owner to, say, develop it for agriculture or extractive resource use, thus reducing the amenity services flow from the area, or to preserve the wilderness. While in principle the owner could charge for access to a wilderness area, in practice this is often infeasible. Further, some of the amenity services that the area delivers do not require access, and cannot be charged for by the owner. The revenue stream that is available under the preservation option is likely to understate the true value to society of that option. This is not true of the development option. In this case, a decision as between the options based on market revenues will be biased in favour of the development option, and the operative question in terms of market failure is whether the existing private

[5] To see the complexities that can arise, note that in some jurisdictions householders may be able to prevent others taking action which reduces the light reaching their property, though this may depend on the nature and purpose of the action.

property rights need to be attenuated, so as to secure the proper, efficient, balance between preservation and development. This sort of issue is dealt with in Part III of the book.

The life-support services provided by the natural environment are not subject to private property rights. Consider, as an example, the global atmosphere, the carbon cycle and the climate system. Historically, the global atmosphere has been a free-access resource. As briefly discussed in Chapter 2, and to be revisited at several places in the rest of the book (especially Chapter 9), anthropogenic emissions of carbon dioxide have increased atmospheric concentrations of that greenhouse gas. The consensus of expert judgement is that this has affected the way that the global climate system works, and that unless action is taken to reduce the rate of growth of anthropogenic carbon dioxide emissions, further change, on balance harmful to human interests, will occur. Given this, most nations are now parties to an international agreement to act to curb the rate of growth of carbon dioxide, and other greenhouse gas, emissions. This agreement is discussed in Chapter 9. It can be seen as a first step in a process of transforming the global atmosphere from a free-access to a common-property resource.

4.9 Public goods

One of the circumstances, 6 in the listing above, required for it to be true that a pure market system could support an efficient allocation is that there be no public goods. Some of the services that the natural environment provides to economic activity have the characteristics of public goods, and cannot be handled properly by a pure market system of economic organisation. So we need to explain what public goods are, the problems that they give rise to for markets, and what can be done about these problems.

4.9.1 What are public goods?

This turns out to be a question to which there is no simple short answer. Public goods have been defined in different ways by different economists. At one time it was thought that there were just private goods and public goods. Now it is recognised that pure private and pure public goods are polar cases, and that a complete classification has to include intermediate cases. It turns out that thinking about these matters helps to clarify some other issues relevant to resource and environmental economics.

There are two characteristics of goods and services that are relevant to the public/private question. These are rivalry and excludability. What we call rivalry is sometimes referred to in the literature as divisibility. Table 4.4 shows the fourfold classification of goods and services that these two characteristics give rise to, and provides an example of each type. Rivalry refers to whether one agent's consumption is at the expense of another's consumption. Excludability refers to whether agents can be prevented from consuming. We use the term 'agent' here as public goods may be things that individuals consume and/or things that firms use as inputs to production. In what follows here we shall generally discuss public goods in terms of things that are of interest to individuals, and it should be kept in mind that similar considerations can arise with some inputs to production.

Pure private goods exhibit both rivalry and excludability. These are 'ordinary' goods and services, the example in Table 4.4 being ice cream. For a given amount of ice cream available, any increase

Table 4.4 Characteristics of private and public goods

	Excludable	Non-excludable
Rivalrous	**Pure private good** Ice cream	**Open-access resource** Ocean fishery (outside territorial waters)
Non-rivalrous	**Congestible resource** Wilderness area	**Pure public good** Defence

in consumption by A must be at the expense of consumption by others, is rival. Any individual can be excluded from ice-cream consumption. Ice cream comes in discrete units, for each of which a consumption entitlement can be identified and traded (or gifted). Pure public goods exhibit neither rivalry nor excludability. The example given is the services of the national defence force. Whatever level that it is provided at is the same for all citizens of the nation. There are no discrete units, entitlement to which can be traded (or gifted). One citizen's consumption is not rival to, at the cost of, that of others, and no citizen can be excluded from consumption.

Open-access natural resources exhibit rivalry but not excludability. The example given in Table 4.4 is an ocean fishery that lies outside of the territorial waters of any nation. In that case, no fishing boat can be prevented from exploiting the fishery, since it is not subject to private property rights and there is no government that has the power to treat it as common property and regulate its exploitation. However, exploitation is definitely rivalrous. An increase in the catch by one fishing boat means that there is less for other boats to take.

Congestible resources exhibit excludability but not, up to the point at which congestion sets in, rivalry. The example given is the services to visitors provided by a wilderness area. If one person visits a wilderness area and consumes its services – recreation, wildlife experiences and solitude, for example – that does not prevent others consuming those services as well. There is no rivalry between the consumption of different individuals, provided that the overall rate of usage is not beyond a threshold level at which congestion occurs in the sense that one individual's visit reduces another's enjoyment of hers. In principle, excludability is possible if the area is either in private ownership or subject to common-property management. In practice, of course, enforcing excludability might be difficult, but, often, given limited points of access to vehicles it is not.

The question of excludability is a matter of law and convention, as well as physical characteristics. We have already noted that as the result of an international agreement that extended states' territorial waters, some ocean fisheries that were open access have become common property. We also noted above that a similar process may be beginning in respect to the global atmosphere, at least in regard to emissions into it of greenhouse gases. In some countries beaches cannot be privately owned, and in some such cases, while beaches actually have the legal status of common property, they are generally used on a free-access basis. This can lead to congestion. In other countries private ownership is the rule, and private owners do restrict access. In some cases where the law enables excludability, on the basis of either private ownership or common property, it is infeasible to enforce it. However, the feasibility of exclusion is a function of technology. The invention of barbed wire and its use in the grazing lands of North America is an historical example. Satellite surveillance could be used to monitor unauthorised use of wilderness areas, though clearly this would be expensive, and presumably at present it is not considered that the benefit from so doing is sufficient to warrant meeting the cost.

In the rest of this section we shall consider pure public goods, which we will refer to simply as 'public goods'. As noted, we will be returning to a detailed consideration of open-access resources, and common-property resources, at several places later in the book. Box 4.2 considers some examples of public goods. Box 4.3 looks at property rights in relation to biodiversity, and the arising implications for incentives regarding conservation and medicinal exploitation.

4.9.2 Public goods and economic efficiency

For our economy with two persons and two private goods, we found that the top-level, product-mix, condition for allocative efficiency was

$$MRUS^A = MRUS^B = MRT \tag{4.14}$$

which is equation 4.8 written slightly differently. As shown in Appendix 4.3, for a two-person economy where X is a public good and Y is a private good, the corresponding top-level condition is:

$$MRUS^A + MRUS^B = MRT \tag{4.15}$$

We have shown that, given certain circumstances, the first of these will be satisfied in a market economy. It follows that the condition which is equation 4.15 will not be satisfied in a market economy. A pure market economy cannot supply public goods at the level required by allocative efficiency criteria.

Box 4.2 Examples of public goods

The classic textbook examples of public goods are lighthouses and national defence systems. These both possess the properties of being non-excludable and non-rival. If you or I choose not to pay for defence or lighthouse services, we cannot be excluded from the benefits of the service, once it is provided to anyone. Moreover, our consumption of the service does not diminish the amount available to others. Bridges also share the property of being non-rival (provided they are not used beyond a point at which congestion effects begin), although they are not typically non-excludable.

Many environmental resources are public goods, as can be seen from the following examples. You should check, in each case, that the key criterion of non-rivalry is satisfied. The benefits from biological diversity, the services of wilderness resources, the climate regulation mechanisms of the earth's atmosphere, and the waste disposal and reprocessing services of environmental sinks all constitute public goods, provided the use made of them is not excessive. Indeed, much public policy towards such environmental resources can be interpreted in terms of regulations or incentives designed to prevent use breaking through such threshold levels.

Some naturally renewing resource systems also share public goods properties. Examples include water resource systems and the composition of the earth's atmosphere. Although in these cases consumption by one person does potentially reduce the amount of the resource available to others (so the resource could be 'scarce' in an economic sense), this will not be relevant in practice as long as consumption rates are low relative to the system's regenerative capacity.

Finally, note that many public health measures, including inoculation and vaccination against infectious diseases, have public goods characteristics, by reducing the probability of any person (whether or not he or she is inoculated or vaccinated) contracting the disease. Similarly, educational and research expenditures are, to some extent, public goods.

Box 4.3 Property rights and biodiversity

Among the many sources of value that humans derive from biological diversity is the contribution it makes to the pharmaceutical industry. This is examined in a volume which brings together a collection of papers on the theme of property rights and biological diversity (Swanson, 1995a). In this box we summarise some of the central issues raised there.

Swanson begins by noting that the biological characteristics of plants (and, to a lesser extent, animals) can be classified into primary and secondary forms. Primary characteristics concern the efficiency with which an organism directly draws upon its environment. For example, plant growth – and the survivability of a population of that plant over time – depends upon its rate of photosynthesis, by which solar energy is converted into the biological material of the plant itself. The success of a species depends on such primary characteristics; indeed, the ecological dominance of humans can be described largely in terms of the massive increases in primary productivity attained through modern agriculture.

But another set of characteristics – secondary characteristics – are also of great importance in the survivability of an organism within its environment. To survive in a particular ecological complex, an organism must be compatible with other living components of its environment. The secondary metabolites which plants develop are crucial in this respect. Some plants develop attractors (such as fruits and aromas) which increase the spread of their reproductive materials. Acorns, for example, are transported and eaten by small animals, thereby encouraging the spread of oak woodlands. Other plants develop repellents in the form of (unattractive) aromas or toxins, which give defence against predatory organisms.

A diverse ecosystem will be characterised by a large variety of biological organisms in which evolutionary processes generate a rich mix of these secondary metabolites. Many of these will be highly context-specific. That is, even within one fairly narrow class of plants, there can be a large variety of these secondary metabolites that function to give relative fitness in a particular

Box 4.3 *continued*

location. These secondary characteristics are helpful to plants and animals not only in aiding current survival but also in terms of long-term evolutionary sustainability. The presence of a diverse collection of secondary metabolites provides resources to help organisms survive environmental disruptions.

But these secondary characteristics are also of immense value to humans, and have been for much of recorded history. Let us look at a few examples discussed by Swanson. Lemons have been used to avoid scurvy in humans for hundreds of years, without any knowledge about how this beneficial effect was taking place. We now know that the active ingredient is vitamin C, one of the secondary metabolites of citrus fruits. Similarly, the bark of the willow tree was used for pain relief for centuries before the active substance (salicylic acid) was identified; its current form is the drug aspirin. More recently, the plant sweetclover was found to be causing severe internal bleeding in cattle. Trials showed that it served as an anti-coagulant across a wide variety of animals. Subsequent developments led to its use in warfarin (the major rodent poison in the world) and in drugs to treat victims of strokes (to reduce blood clotting).

Until recently, almost all medicines were derived more or less directly from natural sources. Even today, in the modern pharmaceuticals industry, a large proportion of the drugs in use throughout the world are derived from natural sources. Much work within the pharmaceuticals industry is concerned with identifying medicinal uses of secondary metabolites within plant, animal and microbial communities. The first step in this process is to develop chemicals from these organisms that have demonstrable biological effects within humans. Possible uses of the chemicals can then be found. What is interesting is that even today, the drugs developed in this way (such as those used in general anaesthesia) are often used without good understanding of their mechanism.

Two things are virtually certain. First, a large number of substances are being, or have been, used in specific cultural contexts without their usefulness having become generally known. Secondly, we have only begun to scratch the surface of the range of possible uses that the biosphere permits. Our collective knowledge encompasses only a small part of what there is to know.

All of this suggests that the conservation of biological diversity is of enormous value. This was recognised in the 1992 Rio Convention on Biological Diversity, which states that biological diversity must be conserved and cultural/institutional diversity respected. Yet the institutional arrangements we have in place are poorly designed to conserve that diversity.

Swanson focuses on the role that property rights plays. The nub of the problem is that the system of property rights which has been built up over the past 100 years rewards the creators of information in very different ways. Consider a drug company that extracts biological specimens from various parts of the world and screens these for potential beneficial effects. Intellectual property rights will be awarded to the first individual or organisation that can demonstrate a novel use of information in a product or process. There is nothing wrong with this, of course. A system which rewards people who create useful information by granting them exclusive rights to market products that incorporate that information is of immense value. Intellectual property rights, in the form of patents and the like, give market value to information, and create incentives to search for and exploit more information.

However, Swanson points out that not all forms of information have such market value. In particular, the existence of biologically diverse ecosystems creates a reservoir of potentially useful information, but no system of property rights exists which rewards those who build up or sustain biodiversity. He writes:

> Internationally-recognised property rights systems must be flexible enough to recognise and reward the contributions to the pharmaceutical industry of each people, irrespective of the nature of the source of that contribution. In particular, if one society generates information useful in the pharmaceutical industry by means of investing in natural capital (non-conversion of forests etc.) whereas another generates such information by investing in human capital (laboratory-based research and school-based training) each is equally entitled to an institution that recognises that contribution.

What is needed, therefore, is a property rights system that brings the value of biodiversity back into human decision making. So-called 'intellectual' property rights should be

Box 4.3 ***continued***

generalised to include not only intellectual but natural sources of information. Put another way, it is *information* property rights rather than just *intellectual* property rights that should be protected and rewarded. An ideal system would reward any investment that generates information, including that which is produced naturally.

It is ironic that the 'success' of modern scientific systems of medicine may be contributing to a loss of potentially useful information. Swanson points to the fact that knowledge which is used with demonstrable success in particular cultural contexts often fails to be widely recognised and rewarded. The difficulty has to do with the fact that this knowledge is not codified in ways that satisfy conventional scientific standards. Publication in academic and professional journals, for example, tends to require analysis in a standard form of each link in the chain running from chemical input to accomplished objective. Unconventional or alternative forms of medicine that cannot fit this pattern struggle to survive, even when they have demonstrable value and where no orthodox substitute exists (such as in the treatment of eczema). Reading the collection of papers in full will show you what Swanson and his co-authors recommend to rectify these shortcomings.

Source: Swanson (1995a, 1995b)

A simple numerical example can provide the rationale for the condition that is equation 4.15. Suppose that an allocation exists such that MRT = 1, $MRUS^A = 1/5$ and $MRUS^B = 2/5$, so that $MRUS^A + MRUS^B < MRT$. The fact that the MRT is 1 means that, at the margin, the private and public commodities can be exchanged in production on a one-for-one basis – the marginal cost of an extra unit of X is a unit of Y, and vice versa. The fact that $MRUS^A$ is 1/5 means that A could suffer a loss of 1 unit of X, and still be as well off if she received 1/5th of a unit of Y by way of compensation. Similarly, the fact that $MRUS^B$ is 2/5 means that B could suffer a loss of 1 unit of X, and still be as well off if she received 2/5 of a unit of Y by way of compensation. Now, consider a reduction in the production of X by 1 unit. Since X is a public good, this means that the consumption of X by both A and B will fall by 1 unit. Given the MRT of 1, the resources released by this reduction in the production of X will produce an extra unit of Y. To remain as well off as initially, A requires 1/5 of a unit of Y and B requires 2/5 of a unit. The total compensation required for both to be as well off as they were initially is 1/5 + 2/5 = 3/5 units of Y, whereas there is available 1 unit of Y. So, at least one of them could actually be made better off than initially, with neither being worse off. This would then be a Pareto improvement. Hence, the initial situation with $MRUS^A + MRUS^B < MRT$ could not have been Pareto optimal, efficient.

Now consider an initial allocation where MRT = 1, $MRUS^A = 2/5$ and $MRUS^B = 4/5$ so that $MRUS^A + MRUS^B > MRT$. Consider an increase of 1 unit in the supply of the public good, so that the consumption of X by both A and B increases by 1 unit. Given MRT = 1, the supply of Y falls by 1 unit. Given $MRUS^A = 2/5$, A could forgo 2/5 units of Y and remain as well off as initially, given X^A increased by 1. Given $MRUS^B = 4/5$, B could forgo 4/5 units of Y and remain as well off as initially, given X^B increased by 1. So with an increase in the supply of X of 1 unit, the supply of Y could be reduced by 2/5 + 4/5 = 6/5 without making either A or B worse off. But, in production the Y cost of an extra unit of X is just 1, which is less than 6/5. So, either A or B could actually be made better off using the 'surplus' Y. For $MRUS^A + MRUS^B > MRT$ there is the possibility of a Pareto improvement, so the initial allocation could not have been efficient.

Since both $MRUS^A + MRUS^B < MRT$ and $MRUS^A + MRUS^B > MRT$ are situations where Pareto improvements are possible, it follows that $MRUS^A + MRUS^B = MRT$ characterises situations where they are not, so it is a necessary condition for allocative efficiency.

In the case of a private good, each individual can consume a different amount. Efficiency requires, however, that all individuals must, at the margin, value it equally. It also requires, see equation 4.14, that the common valuation, at the margin, on the part of individuals is equal to the cost, at the margin, of the good. In the case of a public good, each individual must, by virtue of non-rivalry, consume the same amount of the good. Efficiency does not require that they all value it equally at the margin. It does require, see equation 4.15, that the sum of their marginal valuations is equal to the cost, at the margin, of the good.

Markets cannot provide public goods in the amounts that go with allocative efficiency. In fact, markets cannot supply public goods at all. This follows from their non-excludability characteristic. A market in widgets works on the basis that widget makers exchange the rights to exclusive control over defined bundles of widgets for the rights to exclusive control over defined bundles of something else. Usually, the exchange takes the form of the exchange of widgets for money. This can only work if the widget maker can deny access to widgets to those who do not pay, as is the case with private goods. Where access to widgets is not conditional on payment, a private firm cannot function as it cannot derive revenue from widget production. Given that the direct link between payment and access is broken by non-excludability, goods and services that have that characteristic have to be supplied by some entity that can get the revenue required to cover the costs of production from some source other than the sale of such goods and services. Such an entity is government, which has the power to levy taxes so as to raise revenue. The supply of public goods is (part of) the business of government. The existence of public goods is one of the reasons why all economists see a role for government in economic activity.

Given that it is the government that must supply a public good, the question which naturally arises for an economist is: what rule should government follow so as to supply it in amounts that correspond to efficiency? In principle, the answer to this question follows from equation 4.15. In a two-person, two-commodity economy, the efficient level of supply for the public good is the level at which the sum of two MRUSs is equal to the MRT between it and the private good. Actual economies have many individuals and many private commodities. The first point here presents no difficulty, as it is clear that we simply need to extend the summation over all MRUSs, however many there are. As regards the second, it is simply a matter of noting that the MRT is the marginal cost in terms of forgone private goods consumption, so that the rule becomes: supply the public good at the level where the sum of all the MRUSs is equal to the marginal cost. Now, it follows from its definition that the MRUS is the same as marginal willingness to pay, MWTP, so this rule can be stated as: supply the public good at the level where aggregate marginal willingness to pay is equal to marginal cost. The determination of the efficient amount of a public good, for two individuals for convenience, is illustrated in Figure 4.12.

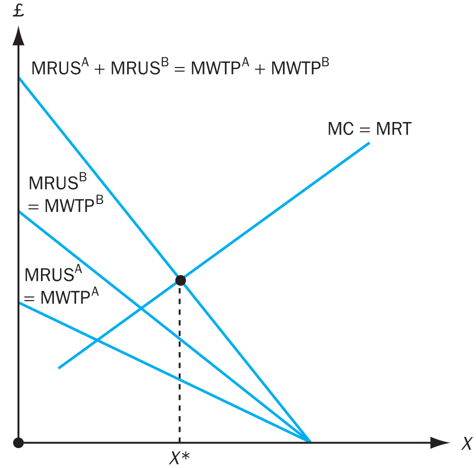


Figure 4.12 The efficient level of supply for a public good

4.9.3 Preference revelation and the free-rider problem

While the rule for the efficient supply of a public good is simple enough at the level of principle, its practical application faces a major difficulty. In order to apply the rule, the government needs to know the preferences, in terms of marginal willingness to pay, of all relevant individuals. It is in the nature of the case that those preferences are not revealed in markets. Further, if the government tries to find out what they are in some other way, then individuals have (on the standard assumptions about their motivations and behaviour) incentives not to reveal their preferences truthfully. Given that all consume equal amounts of a public good, and that exclusion from consumption on account of non-payment is impossible, individuals will try to 'free-ride' with respect to public goods provision.

To bring out the basic ideas here in a simple way we shall consider an example where the problem is to decide whether or not to provide a discrete amount of a public good, rather than to decide how much of a public good to supply. The nature of the problem is the same in either case, but is easier to state and understand in the 'yes/no?' case than in the 'how much?' case. At issue is the question of whether or not to install street lighting. We will first look at this when there is no government. There are two individuals A and B. Both have an endowment of private goods worth £1000. Installing the street lighting will cost £100. The two individuals both have preferences such that they would be willing to pay £60 for the installation of street lighting. The analysis that follows is not dependent on the two individuals being equally well off and having the same preferences. That just makes the story easier to tell initially. An obvious modification of the rule derived for the efficient level of provision of a public good derived above for the 'yes/no' situation is that the decision should be 'yes' if the sum of individuals' willingness to pay is equal to or greater than the cost. In this case it is greater – £60 + £60 = £120.

Now, suppose that A and B agree to proceed in the following way. Each will independently write down on a piece of paper either 'Buy' or 'Don't Buy'. If when the two pieces of paper are brought together, both have said 'Buy', they buy the street lighting jointly and share the cost equally. For two 'Don't Buy' responses, the street lighting is not bought and installed. In the event of one 'Buy' and one 'Don't Buy', the street lighting is bought and the individual who voted 'Buy' pays the entire cost. The four possible outcomes are shown in the cells of Table 4.5 in terms of the monetary valuations on the part of each individual, that of A to the left of the slash, that of B to the right.[6]

Table 4.5 The preference revelation problem

		B	
		Buy	Don't buy
A	Buy	1010/1010	960/1060
	Don't buy	1060/960	1000/1000

In the bottom-right cell, the decision is not to go ahead. Neither incurs any cost in regard to street lighting and neither gets any benefit, so both are in their initial situations with £1000. Suppose both responded 'Buy'. Then with the street lighting installed, as shown in the top-left cell, the situation for both can be expressed in monetary terms as £1010. Each has paid £50, half of the total of £100, for something valued at £60, so gaining by £10 as compared with the no street lighting situation. Suppose A wrote 'Buy' and B wrote 'Don't Buy'. The lighting goes in, A pays the whole cost and B pays nothing. A pays £100 for something she values at £60, and goes from £1000 to £960. B pays nothing for something she values at £60, and goes from £1000 to £1060. This is shown in the top-right cell. The bottom-left cell has the entries of that cell reversed, because B pays the whole cost.

Now, clearly both are better off if both write 'Yes' and the street lighting is bought. But, either will be even better off if, as in the bottom left or top right cell, she can 'free ride'. For each individual thinking about what to write on her piece of paper, writing 'Don't Buy' is the dominant strategy. Consider individual B. If A goes for 'Buy', B gets to £1010 for 'Buy' and to £1060 for 'Don't Buy'. If A goes for 'Don't Buy', B gets to £960 for 'Buy' and to £1000 for 'Don't Buy'. Whatever A does, B is better off going for 'Don't Buy'. And the same is true for A, as can readily be checked. So, while installing the lighting and sharing the cost equally is a Pareto improvement, it will not come about where

[6] This is a 'game' with the structure often referred to as the 'Prisoner's Dilemma' because of the setting in which the structure is often articulated. A 'game' is a situation in which agents have to take decisions the consequences of which depend on the decisions of other agents. We shall come back to looking at some game structures in Chapter 9. In the Prisoner's Dilemma setting, the agents are two individuals arrested for a crime and subsequently kept apart so that they cannot communicate with one another. The evidence against them is weak, and the police offer each a deal – confess to the crime and get a much lighter sentence than if you are convicted without confessing. Confession by one implicates the other. If neither confesses, both go free. If both confess, both get lighter sentences. If only one confesses, the confessor gets a light sentence while the other gets a heavy sentence. The dominant strategy is confession, though both would be better off not confessing.

both individuals act independently to serve their own self-interest. What is needed is some kind of coordination, so as to bring about the Pareto improvement which is going ahead with the street lighting.

Given what we have already said about public goods, government would seem the obvious way to bring about the required coordination. It can, in principle, ascertain whether the installation of street lighting is justified on efficiency grounds, and if it is, install it and cover the cost by taxing each individual according to their willingness to pay. However, in practice, given self-seeking individuals, the free-rider problem also attends this programme. The problem comes up in trying to get the individuals to reveal their true preferences for the public good.

Suppose now that a government does exist, and that it wants to follow efficiency criteria. It knows that installing the street lighting will cost £100, and that it should install it if total willingness to pay is equal to or greater than that. It does not know the preferences, in terms of willingness to pay, of the two individuals who, in this simple example, comprise the citizenry. The obvious thing for it to do is to ask them about it. It does that, stating that the cost of installation will be met by a tax on each individual which is proportional to her willingness to pay and such that the total tax raised is equal to the cost of installation. If each individual truly reports willingness to pay of £60, the street lighting will go ahead and each will pay £50 in tax. This represents a Pareto improvement – see the top left cell in Table 4.5. The problem is that the incentives facing each individual are not such as to guarantee truthful preference revelation. Given that tax liability will be proportional to stated willingness to pay, there is an incentive to understate it so as to reduce the tax liability if the street lighting goes ahead, and to get something of a free ride. In the example of Table 4.5, if B states willingness to pay as £40 and A tells the truth, the street lighting will go ahead – stated aggregate willingness to pay £100 – and B will pay 40%, rather than 50%, of £100. If A also understates her willingness to pay by £20, the government's estimate of aggregate willingness to pay will mean that it does not go ahead with the lighting. The attempt to free-ride may fail if many make it.

The problem of securing truthful preference revelation in regard to the supply of public goods has been the subject of a lot of investigation by economists. It turns out to be very difficult to come up with systems that provide the incentives for truthful revelation, and are feasible. The interested reader will find references to work in this area in the Further Reading section at the end of the chapter. Here we will, in order to indicate the nature of the difficulties, simply note one idea that is intended to overcome the free-riding incentives generated by the system just discussed. There the problem was that an individual's tax liability depended on stated willingness to pay. This could be avoided by the government asking about willingness to pay on the understanding that each individual would, if the installation went ahead, pay a fixed sum. Suppose that the government divided the cost by the number of individuals, and stated that the fixed sum was £50 per individual. For both individuals, true willingness to pay is £60. Both have an incentive now to overstate their willingness to pay. Both value the street lighting at more than it is going to cost them so they want to see it installed. Both know that this is more likely the higher they say that their willingness to pay is, and that however much in excess of £60 they report they will only pay £50.

In this case overstating willingness to pay produces the right decision. The street lighting should be installed on the basis of true aggregate willingness to pay, and will be installed on the basis of reported willingness to pay. If the lighting is installed, each individual is better off, there is a Pareto improvement. Suppose, however, that A's willingness to pay is £55 and B's is £40. In that case, aggregate willingness to pay is £95, less than the cost of £100, and the street lighting should not be installed. In this case, on the understanding that each would pay a tax of £50 if the lighting is installed, A would have an incentive to overstate her willingness to pay as before, but B would have an incentive to understate hers. In fact, it would make sense for B to report willingness to pay as £0 – if the lighting goes ahead she pays £50 for something worth just £40 to her, so she will want to do the most she can to stop it going ahead. Whether it does go ahead or not depends on how much A overstates her willingness to pay by. If A reports £200 or more, despite B reporting £0, the street lighting will be installed when on efficiency grounds it should not be.

Finally, this simple example can be used to show that even if the government could secure the truthful revelation of preferences, public goods supply is still a difficult problem. Suppose that A's true willingness to pay is £60 and B's is £41, and that somehow or other the government knows this without needing to ask the individuals. The government has to decide how to cover the cost. It could tax each in proportion to willingness to pay, but given that A and B are initially equally wealthy in terms of private goods, this is in practice unlikely as it would be regarded as unfair. Taxing each at equal shares of the cost would likely be seen as the 'fair' thing to do. In that case, A would pay £50 for a benefit worth £60, and B would pay £50 for a benefit worth £41. In monetary terms, as the result of installing the lighting, A would go from £1000 to £1010 and B would go from £1000 to £991. Since there is a loser this is not a Pareto improvement, though it is a potential Pareto improvement – we are into the domain of the Kaldor–Hicks–Scitovsky test. By looking at equally wealthy individuals, we avoided the problem that efficiency gains are not necessarily welfare gains. Suppose that the gainer A were much richer than the loser B. Then, the question arises as to whether gains and losses should be given equal weight in coming to a decision.

For a government to make decisions about the supply and financing of public goods according to the criteria recommended by economists requires that it has lots of difficult-to-acquire information, and can involve equity questions as well as efficiency questions.

4.10 Externalities

An external effect, or an externality, is said to occur when the production or consumption decisions of one agent have an impact on the utility or profit of another agent in an unintended way, and when no compensation/payment is made by the generator of the impact to the affected party.[7] In our analysis thus far in this chapter, we have excluded the existence of externalities by the assumptions that were made about the utility and production functions. But in practice consumption and production behaviour by some agents does affect, in uncompensated/unpaid for ways, the utility gained by other consumers and the output produced, and profit realised, by other producers. Economic behaviour does, in fact, involve external effects.

The stated definition of an external effect is not perhaps very illuminating as to what exactly is involved. Things will become clearer as we work through the analysis. The two key things to keep in mind are that we are interested in effects from one agent to another which are unintended, and where there is no compensation, in respect of a harmful effect, or payment, in respect of a beneficial effect. We begin our analysis of externalities by discussing the forms that externalities can take.

4.10.1 Classification of externalities

In our two-person, two-(private)-commodity, two-input economy we have worked with

$$U^A = U^A(X^A, Y^A)$$

$$U^B = U^B(X^B, Y^B)$$

as utility functions, and

$$X = X(K^X, L^X)$$

$$Y = Y(K^Y, L^Y)$$

as production functions. Note that here the only things that affect an individual's utility are her own consumption levels, and that the only things that affect a firm's output are the levels of inputs that it uses. There are, that is, no external effects.

[7] Some authors leave out from the definition of an externality the condition that the effect is not paid or compensated for, on the grounds that if there were payment or compensation then there would be no lack of intention involved, so that the lack of compensation/payment part of the definition as given in the text here is redundant. As we shall see, there is something in this. However, we prefer the definition given here as it calls attention to the fact that lack of compensation/payment is a key feature of externality as a policy problem. Policy solutions to externality problems always involve introducing some kind of compensation/payment so as to remove the unintentionality, though it has to be said that the compensation/payment does not necessarily go to/come from the affected agent.

External effects can, first, be classified according to what sort of economic activity they originate in and what sort of economic activity they impact on. Given two sorts of economic activity, consumption and production, this gives rise to the sixfold classification shown in Table 4.6. The first column shows whether the originating agent is a consumer or producer, the second whether the affected agent is a consumer or producer, and the third provides an illustrative utility or production function for the affected agent. In Table 4.6, we are concerned only to set out the forms that unintended interdependence between agents could take. Some examples will be provided shortly.

Table 4.6 Externality classification

Arising in	Affecting	Utility/production function
Consumption	Consumption	$U^A(X^A, Y^A, X^B)$
Consumption	Production	$X(K^X, L^X, Y^A)$
Consumption	Consumption and production	$U^A(X^A, Y^A, X^B)$ and $Y(K^Y, L^Y, X^B)$
Production	Consumption	$U^A(X^A, Y^A, X)$
Production	Production	$X(K^X, L^X, Y)$
Production	Consumption and production	$U^A(X^A, Y^A, Y)$ and $X(K^X, L^X, Y)$

In the first row in Table 4.6, an example of a consumption externality is where agent B's consumption of commodity X is an argument in A's utility function – B's consumption of X affects the utility that A derives from given levels of consumption of X and Y. In our discussion of this type of externality in Section 4.10.3, we follow most of the economics literature in looking at a situation where something that A does is something that B does not like. A smoking is a very common example used in the literature. We use A playing a musical instrument loudly. However, we should note that the literature on the determinants of self-assessed happiness that we briefly looked at in Chapter 3 (see Section 3.3.4 and references in Further Reading at the end of that chapter) suggests that consumption-to-consumption externalities where it is B's consumption of ordinary commodities that A also consumes, as with X in Table 4.6, are likely to be common. This is rivalry, or the 'keeping up with the Joneses' effect.

In the second row, A's consumption of Y is shown as affecting the production of X, for given levels of capital and labour input. Row three has B's consumption of X affecting both A's utility and the production of Y. In row 4, the amount of X produced, as well as A's consumption of X, affects A's utility. Row 5 has the production of Y determining, for given capital and labour inputs, the amount of X produced. Finally, in row 6 we have a situation where the level of Y affects both A's utility and the production of X.

The unintended impact that an external effect involves may be harmful or beneficial. Table 4.7 provides examples of both kinds. If an individual has a vaccination, it protects her, which is her intention, but it also has the unintended effect of reducing the probability that others will contract the disease. An individual playing her radio loudly in the park inflicts suffering on others, though that is not her intention. In these two cases, the external effect originates in consumption and affects individuals. A beneficial externality originating in production, and impacting on production, is the case where a honey-producer's bees pollinate a nearby fruit orchard. Pollution, in the bottom right cell, is a harmful externality which most usually originates in production activities. It can affect consumers, or producers, or both.

Another dimension according to which external effects can be classified is in terms of whether they have, or do not have, the public goods characteristics of non-rivalry and non-excludability. While external effects can have the characteristics of private goods,

Table 4.7 Beneficial and harmful externalities

Effect on others	Originating in consumption	Originating in production
Beneficial	Vaccination against an infectious disease	Pollination of blossom arising from proximity to apiary
Adverse	Noise pollution from radio playing in park	Chemical factory discharge of contaminated water into water systems

those that are most relevant for policy analysis exhibit non-rivalry and non-excludability. This is especially the case with external effects that involve the natural environment, which mainly involve pollution problems. Why this is the case will become clear in the analysis that follows here. All of the examples in Table 4.7 involve non-rivalry and non-excludability.

4.10.2 Externalities and economic efficiency

Externalities are a source of market failure. Given that all of the other institutional conditions for a pure market system to realise an efficient allocation hold, if there is a beneficial externality the market will produce too little of it in relation to the requirements of allocative efficiency, while in the case of a harmful externality the market will produce more of it than efficiency requires. Since we are concerned with the application of welfare economics to environmental problems, and the main relevance of externalities there is in regard to environmental pollution, we shall look in any detail only at harmful externalities here. Box 4.4 concerns an important example of a harmful externality pollution problem. We will demonstrate that the market, in the absence of corrective policy, will 'over-supply' pollution by looking at three sorts of pollution problem – a consumer-to-consumer case, a producer-to-producer case, and a case where the unintended effect is from a producer to consumers. These three cases bring out all of the essential features of pollution as a market failure problem. In the text we shall use diagrams and partial equilibrium analysis to make the essential points – the reader may find it useful to review our exposition of this method of analysis earlier in this chapter. In Appendix 4.3 we cover the same ground using general equilibrium analysis.

Before getting into these cases in a little detail, we can make a general intuitive point that covers both beneficial and harmful externalities. The basic problem with external effects follows directly from the definition in regard to unintendedness and lack of payment/compensation. These two features of the externality problem are directly related. The lack of intentionality follows from the fact that the impact involved does not carry with it any recompense, in

Box 4.4 Atmospheric ozone and market failure

Evidence now suggests that the accumulation of tropospheric ozone in urban areas poses serious threats to human health, and also leads to agricultural crop damage in surrounding areas. This accumulation of ozone in lower layers of the atmosphere is distinct from the destruction of the ozone layer in the earth's upper atmosphere (the stratosphere), which phenomenon – often known as 'holes in the ozone layer' – causes different problems, as is explained in Chapter 9.

A major source of tropospheric ozone is road vehicle exhaust emissions. Because vehicle emissions have real effects on well-being through our utility and production functions, these emissions can be termed 'goods' (although it may be preferable to label them as 'bads' as the effects on utility are adverse). However, with no individual private property rights in clean air, in the absence of government intervention, no charge is made for such emissions. The analysis below will demonstrate that, as a result, resources are not being allocated efficiently. An efficient allocation would involve lower exhaust emissions, implying one or more of: lower traffic volumes, change in fuel type used, increased engine efficiency, enhanced exhaust control. How such objectives might be achieved is considered in this chapter, and in more detail in Chapters 7 and 16, but it should be clear at this stage that one method would be through the use of a tax on the emissions that cause ozone accumulation. An efficient emissions tax would impose a tax rate equal to the value of the marginal damage that would occur at the efficient level of emissions.

In arriving at this conclusion, we do not explicitly consider the time dimension of pollution. But note that if ozone accumulates over time, and damage is dependent on the stock of ozone rather than the flow of emissions in any particular period, then we need to consider the accumulation of the pollutant over time. As Chapter 16 shows, where emission flows lead to accumulating stocks of pollutants, it may be efficient to impose a tax rate that rises over time.

the case of a beneficial effect, or penalty, in the case of a harmful effect. External effects arise where an agent's actions affecting other agents do not involve any feedback – benefit is conferred which is not rewarded, or harm is done which is not punished. Given the lack of reward/punishment, which in a market system would be signalled by monetary payment, an agent will not take any account of the effect concerned. It will be unintended and 'external' to her decision making. Where it is a beneficial effect, it will not be encouraged sufficiently, and there will not be enough of it. Where it is a harmful effect, it will not be discouraged sufficiently, and there will be too much of it. The key to dealing with the market failure that external effects give rise to is to put in place the missing feedbacks, to create a system which does reward/punish the generation of beneficial/harmful effects, so that they are no longer unintentional.

4.10.3 Consumption–consumption externality

Suppose that A and B live in adjacent flats (apartments). A is a saxophone player, who enjoys practising a lot. B does not like music, and can hear A practising. The utility functions are

$$U^A = U^A(M^A, S^A)$$

$$U^B = U^B(M^B, S^A)$$

where M represents wealth and S^A is the hours that A plays the saxophone each week, with $\partial U^A/\partial M^A > 0$, $\partial U^B/\partial M^B > 0$, $\partial U^A/\partial S^A > 0$ and $\partial U^B/\partial S^A < 0$. In Figure 4.13 we show, as MB, the marginal benefit of playing to A, and, as MEC for marginal external cost, the marginal cost of playing to B. Marginal benefit is the amount that A would pay, if it were necessary, to play a little more. Conversely, MB is the amount of compensation that would be required to leave A as well off given a small reduction in playing. Marginal external cost is the amount that B would be willing to pay for a little less playing. Conversely, MEC is the amount of compensation that would be required to leave B as well off given a small increase in S (hours of A's saxophone playing).

Given that A does not in fact have to pay anything to play her saxophone in her flat, she will increase her hours of playing up to the level S_0, where MB is

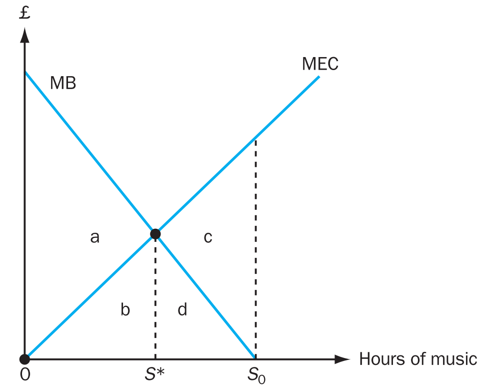


Figure 4.13 The bargaining solution to an externality problem

equal to zero. At that level, A's total benefit from playing is given by the sum of the areas of the triangles a, b and d, and B's total suffering is measured in money terms by the sum of the areas b, d and c.

This is not an efficient outcome, because at S_0 MEC > MB. The efficient outcome is at S^* where MEC = MB. At any S to the left of S^* MB > MEC so that for a small increase in S A would be willing to pay more than would compensate B for that increase. At any S to the right of S^* MEC > MB so that for a small decrease in S B would be willing to pay more for a small decrease in S than would be required to compensate A for that decrease. The inefficient level of saxophone playing at S_0 comes about because there are no payments in respect of variations in S, no market in S, so that the effect on B is unintentional on the part of A.

At the level of principle, the solution to this problem of inefficiency is fairly obvious. The problem is that A does not compensate B because B does not have any legal right to such compensation, does not have a property right in a domestic environment unpolluted by saxophone music. So, the solution is to establish such a property right, to give B the legal right to a domestic environment that is not noise polluted. Such legal arrangements would support bargaining which would lead to S^* as the level of S. The argument that establishes that S^* would be the outcome under a legal regime where B can claim compensation from A exactly parallels the argument that establishes that S^* is the efficient outcome. To the left of S^*, with MB > MEC, A will be willing to pay more in compensation for a small increase in S than B requires, so will pay and play more. A will not increase S beyond S^* because the compensation that it would be necessary to pay B would be greater than the worth to A of the small increase thereby attained.

4.10.3.1 The Coase theorem

The idea that, given a suitable assignment of property rights, private bargaining between individuals can correct externality problems and lead to efficient outcomes is generally attributed to the Nobel prize-winning economist Ronald Coase, and the result discussed above is often referred to as the 'Coase theorem' (the seminal paper is Coase, 1960). In fact, the result discussed above is only half of the Coase theorem. The other half says that an efficient outcome can also be attained by vesting the property right in the generator of the external effect. In that case, the generator would have the legal right to play, for this example, as much saxophone as she liked. The point is that given that right, it could be in the interests of the victim to offer money to the generator not to exercise her right to the full. Just as the absence of a clear property right vested in the victim inhibits one kind of bargaining, so does the absence of a clear property right vested in the generator inhibit another kind of bargaining.

Suppose then, that in our saxophone-playing example a law is passed saying that all saxophone players have an absolute right to practise up to the limits of their physical endurance. Legally A can play as much as she wants. But, a legal right can be traded. So, the opportunity now exists for A and B to bargain to a contract specifying the amount that A will actually play. That amount will be S^* in Figure 4.13. To the right of S^* MEC > MB so B's willingness to pay for a small reduction is greater than the compensation that A requires for that small reduction. Starting at S_0 and considering successive small reductions, B will be offering more than A requires until S^* is reached where B's offer will exactly match the least that A would accept. A and B would not be able to agree on a level of S to the left of S^*, since there B's willingness to pay is less than A requires by way of compensation.

So, what the Coase theorem actually says is that given this kind of externality situation, due to incomplete private property rights, one solution involves creating property rights for either the victim or the generator, and that either assignment will lead to an efficient outcome. It needs to be explicitly and carefully noted here that there are two things that are not being claimed. First, that it is not being said that the outcome will be the same in both cases. Second, that it is not being said that either way of assigning property rights necessarily promotes equity.

In regard to the first point here, note that considering the move from S_0 to S^* in our saxophone music example consequent upon the establishment of the property right and the ensuing bargaining we have:

a. For the case where B gets the property right – there is an S reduction of $(S_0 - S^*)$ and A pays B an amount equal to the area of triangle b, the money value of B's suffering at the efficient outcome S^*.
b. For the case where A gets the property right – there is an S reduction of $(S_0 - S^*)$ and B pays A an amount equal to the area of triangle d, the money value of A's loss as compared with the no-property-rights situation.

Clearly, which way the property right is assigned affects the wealth of A or B. To be granted a new property right is to have one's potential monetary wealth increased. In case a, B experiences fewer saxophone hours and an increase in wealth by virtue of a payment from A, so that A's wealth goes down with her pleasure from playing. In case b, B experiences fewer saxophone hours and a decrease in wealth by virtue of a payment to A, who gets less pleasure from playing. As we have drawn Figure 4.13, in neither case a nor b does the increase in wealth affect the receiving individual's tastes. In case a, that is, B's willingness to pay for fewer music hours is not affected by becoming wealthier – the slope of the MEC line does not change. In case b, A's willingness to pay for more music hours is not affected by becoming wealthier – the slope and position of the MB line do not change. While these assumptions may be plausible in this example, they clearly are not generally appropriate. They were imposed here to produce a simple and clear graphical representation. If the assumption that tastes are unaffected by wealth increases is dropped, then with the case a assignment MEC would shift and with the case b assignment MB would shift. In neither case then would S^* as shown in Figure 4.13 be the bargaining outcome, and the outcomes would be different in the two cases. Both outcomes would be efficient, because in both cases we would have MB = MEC, but they would involve different levels of S.

So, the first point is that the Coase theorem properly understood says that there will be an efficient outcome under either assignment of property rights, not that there will be the same efficient outcome under either assignment. The second point, concerning equity, is simply that there is no implication that either assignment will have any desirable implications in terms of equity. This follows directly from our earlier discussions of the relationship between optimality and efficiency. In the case of our saxophone example, we have said nothing about the initial wealth/income situations of the two individuals. Clearly, our views on which way the property right should be assigned will, unless we are totally uninterested in equity, be affected by the wealth/income of the two individuals. Given that efficiency criteria do not discriminate between the two possible assignments of property rights, it might seem natural to take the view that the assignment should be on the basis of equity considerations. Unfortunately, this does not lead to any generally applicable rules. It is not always the case that externality sufferers are relatively poor and generators relatively rich, or vice versa. Even if we confine attention to a particular class of nuisance, such as saxophone playing in flats, it cannot be presumed that sufferers deserve, on equity grounds, to get the property right – some may be poor in relation to their neighbour and some rich.

Given the simple and compelling logic of the arguments of the Coase theorem, the question arises as to why uncorrected externalities are a problem. If they exist by virtue of poorly defined property rights and can be solved by the assignment of clearly defined property rights, why have legislatures not acted to deal with externality problems by assigning property rights? A full answer to this question would be well beyond the scope of this book, but the following points are worthy of note. First, as we have seen, the case for property rights solutions is entirely an efficiency case. Legislators do not give efficiency criteria the weight that economists do – they are interested in all sorts of other criteria. Second, even given clearly defined property rights, bargaining is costly. The costs increase with the number of participants. While expositions of the Coase theorem deal with small numbers of generators and sufferers, typically one of each, externality problems that are matters for serious policy concern generally involve many generators and/or many sufferers, and are often such that it is difficult and expensive to relate one particular agent's suffering to another particular agent's action. This makes bargaining expensive, even if the necessary property rights exist in law. The costs of bargaining, or more generally 'transactions costs', may be so great as to make bargaining infeasible. Third, even leaving aside the large numbers problem, in many cases of interest the externality has public bad characteristics which preclude bargaining as a solution.[8] We shall discuss this last point in the context of producer to consumer externalities.

4.10.4 Production–production externality

For situations where numbers are small, this case can be dealt with rather quickly. Consider two firms with production functions

$$X = X(K^X, L^X, S)$$

$$Y = Y(K^Y, L^Y, S)$$

where S stands for pollutant emissions arising in the production of Y, which emissions affect the output of X for given levels of K and L input there. As an example, Y is paper produced in a mill which discharges effluent S into a river upstream from a laundry which extracts water from the river to produce clean linen, X. Then, the assumption is that $\partial Y/\partial S > 0$, so that for given levels of K^Y and L^Y lower S emissions means lower Y output, and that $\partial X/\partial S < 0$, so that for given levels of K^X and L^X higher S means lower X.[9]

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[8] 'Public bad' is a term often used for a public good that confers negative, rather than positive, utility on those affected by it.

[9] Note that we are guilty here of something that we cautioned against in Chapter 2 in our discussion of the materials balance principle – writing a production function in which there is a material output, S, for which no material input basis is given. We do this in the interests of simplicity. A more appropriate production function specification is given in Appendix 4.3, where it is shown that the essential point for present purposes is not affected by our shortcut in the interests of simplicity.

This externality situation is amenable to exactly the same kind of treatment as the consumer-to-consumer case just considered. Property rights could be assigned to the downstream sufferer or to the upstream generator. Bargaining could then, in either case, produce an efficient outcome. To see this simply requires the reinterpretation of the horizontal axis in Figure 4.13 so that it refers to S as pollutant emissions. For profits in the production of X we have

$$\pi^X = P_X X(K^X, L^X, S) - P_K K^X - P_L L^X$$

where $\partial\pi^X/\partial S < 0$. The impact of a small increase in S on profits in the production of X is, in the terminology of Figure 4.13, marginal external cost, MEC. For profits in the production of Y we have

$$\pi^Y = P_Y Y(K^Y, L^Y, S) - P_K K^Y - P_L L^Y$$

where $\partial\pi^Y/\partial S > 0$. The impact of a small increase in S on profits in the production of Y is, in the terminology of Figure 4.13, marginal benefit, MB. With these reinterpretations, the previous analysis using Figure 4.13 applies to the producer-to-producer case – in the absence of a well-defined property right S will be too large for efficiency, while an efficient outcome can result from bargaining based on a property right assigned to either the producer of X or the producer of Y.

An alternative way of internalising the externality would be to have the firms collude so as to maximise their joint profits. That this would produce an efficient outcome is proved in Appendix 4.3. The matter is, however, quite intuitive. The externality arises because the Y producer does not take account of the effects of its actions on the output for given inputs of the X producer. If the Y producer chooses its levels of K^Y, L^Y and S in the light of the consequences for the output of X for given K^X and L^X, and hence on the profits arising in the production of X, then those consequences will not be unintended. On the contrary, the two firms will be operated as if they were a single firm producing two commodities. We know that a single firm producing a single commodity will behave as required for efficiency, given all of the ideal conditions. All that is being said now is that this result carries over to a firm producing two commodities. For the firm that is producing both X and Y the ideal conditions do apply, as there is no impact on its activities the level of which is unintentionally set by others.

While joint profit maximisation can internalise an externality as required for efficiency, there appear to be few, if any, recorded instances of firms colluding, or merging, so as to internalise a pollution externality. Collusion to maximise joint profits will occur only if both firms believe that their share of maximised joint profits will be larger than the profits earned separately. There is, in general, no reason to suppose that cases where there is the prospect of both firms making higher profits with collusion will coincide with circumstances where there is a recognised inter-firm pollution externality.

4.10.5 Production–consumption externality

The key feature of the case to be considered now is that the external effect impacts on two agents, and with respect to them is non-rival and non-excludable in consumption. As is the case generally in this chapter, 'two' is a convenient way of looking at 'many' – the two case brings out all the essential features of the many case while simplifying the notation and the analysis. Putting this key feature in the context of the production-to-consumption case aligns with the perceived nature of the pollution problems seen as most relevant to policy determination. These are typically seen as being situations where emissions arising in production adversely affect many individuals in ways that are non-rival and non-excludable.

So, in terms of our two-person, two-commodity economy we assume that:

$$U^A = U^A(X^A, Y^A, S) \text{ with } \partial U^A/\partial S < 0$$

$$U^B = U^B(X^B, Y^B, \mathrm{S}) \text{ with } \partial U^B/\partial S < 0$$

$$X = X(K^X, L^X)$$

$$Y = Y(K^Y, L^Y, S) \text{ with } \partial Y/\partial S > 0$$

Emissions arise in the production of Y and adversely affect the utilities of A and B. The pollution experienced by A and B is non-rival and non-excludable. A concrete example, bearing in mind that 'two' stands for 'many', would be a fossil-fuel-burning electricity plant located in an urban area. Its emissions pollute the urban airshed, and, to a first approximation, all who live within the affected area experience the same level of atmospheric pollution.

Given our earlier discussion of the supply of public goods, we can immediately conclude here that private bargaining based on some assignment of property rights will not deal with the externality problem. And, the joint profit maximisation solution is not relevant. In this kind of situation, correcting the market failure requires some kind of ongoing intervention in the workings of the market by some government agency. As we shall consider at some length and in some detail in Part II of the book, there is a range of means of intervention that the government agency, call it an Environmental Protection Agency or EPA, could use. Here, we shall just look at the use of taxation by the EPA, so as to bring out the essential features of the situation where the externality has the characteristics of a public bad. A formal general equilibrium analysis is sketched in Appendix 4.3. Here we shall use partial equilibrium analysis based on Figure 4.14.

It introduces some new terminology. PMC stands for private marginal cost. Private costs are the input costs that the Y producer actually takes account of in determining its profit-maximising output level, i.e.

$$C = P_K K^Y + P_L L^Y = C(Y)$$

so that PMC = ∂C/∂Y. We introduced the idea of MEC (marginal external cost) in considering the consumer-to-consumer case, as the amount that the sufferer would be willing to pay to reduce suffering by a small amount. In the present case there are two sufferers and MEC is the sum of the willingness to pay of each of them, as consumption of suffering is non-rival and non-excludable. We define social marginal cost as:

$$\text{SMC} = \text{PMC} + \text{MEC}$$

Figure 4.14 shows PMC increasing with Y in the usual way. The SMC line has a steeper slope than the PMC line, so that MEC is increasing with Y – as Y production increases, S output increases.

To maximise profit, the Y firm will produce at Y_0, where PMC is equal to the output price P_Y. This is not the Y output that goes with efficiency, as in balancing costs and benefits at the margin it is ignoring the costs borne by A and B. Efficiency requires the balancing at the margin of benefits and costs which include the external costs borne by A and B. The efficient output level for Y is, that is, Y^* where SMC equals P_Y. In the absence of any correction of the market failure that is the external costs imposed on A and B, the market-determined level of Y output will be too high for efficiency, as will the corresponding level of S be.

To correct this market failure the EPA can tax S at a suitable rate. In Figure 4.14, we show a line labelled PMCT, which stands for private marginal cost with the tax in place. This line shows how the Y firm's marginal costs behave given that the EPA is taxing S at the appropriate rate. As shown in Figure 4.14, the appropriate tax rate is

$$t = \text{SMC}^* - \text{PMC}^* = \text{MEC}^* \tag{4.16}$$

i.e. the tax needs to be equal to marginal external cost at the efficient levels of Y and S. In Appendix 4.3 we show that another way of stating this is:

$$t = P_X[\text{MRUS}^{\text{A}}_{XS} + \text{MRUS}^{\text{B}}_{XS}] \tag{4.17}$$

Comparing equations 4.16 and 4.17, we are saying that

$$\text{MEC}^* = P_X[\text{MRUS}^{\text{A}}_{XS} + \text{MRUS}^{\text{B}}_{XS}] \tag{4.18}$$

This makes a lot of sense. Recall that MRUS stands for marginal rate of utility substitution. The XS subscripts indicate that it is the MRUS for commodity X and pollution S that is involved here. Recall also that the MRUS gives the amount of the increase in, in this case, X that would keep utility constant in the face of a small increase in S. Equation 4.18 says that

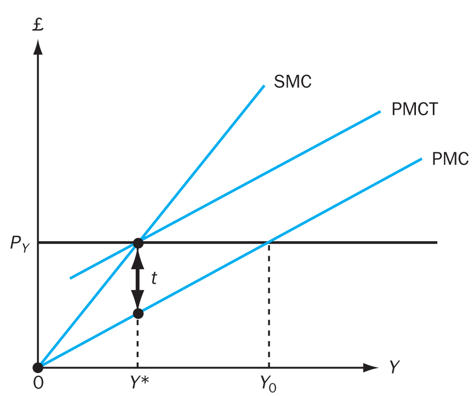

Figure 4.14 Taxation for externality correction

MEC* is the monetary value of the extra consumption of commodity X by A and B that would be required to compensate them both for a small increase in S, from the efficient level of S. In saying this we are choosing to use the commodity X as the compensation vehicle. We could equally well have chosen the commodity Y for this purpose and derived

$$t = P_Y[\text{MRUS}^{\text{A}}_{YS} + \text{MRUS}^{\text{B}}_{YS}] \tag{4.19.a}$$

and

$$\text{MEC*} = P_Y[\text{MRUS}^{\text{A}}_{YS} + \text{MRUS}^{\text{B}}_{YS}] \tag{4.19.b}$$

Taxation at the rate MEC* is required to bring about efficiency. Note that the tax rate required is not MEC at Y_0, is not MEC in the uncorrected situation. In order to be able to impose taxation of emissions at the required rate, the EPA would need to be able to identify Y^*. Given that prior to EPA intervention what is actually happening is Y_0, identification of Y^* and calculation of the corresponding MEC* would require that the EPA knew how MEC varied with S, i.e. knew the utility functions of A and B. It is in the nature of the case that this information is not revealed in markets. The problems of preference revelation in regard to public goods were discussed above. Clearly, those problems carry over to public bads such as pollution. The implications of this for feasible policy in respect of pollution control by taxation are discussed in Part II of the book.

Finally here we should note that the basic nature of the result derived here for the case where just one production activity gives rise to the emissions of concern carries over to the case where the emissions arise in more than one production activity. Consider a two-person, two-commodity economy where:

$$U^{\text{A}} = U^{\text{A}}(X^{\text{A}}, Y^{\text{A}}, S) \text{ with } \partial U^{\text{A}}/\partial S < 0$$

$$U^{\text{B}} = U^{\text{B}}(X^{\text{B}}, Y^{\text{B}}, S) \text{ with } \partial U^{\text{B}}/\partial S < 0$$

$$X = X(K^X, L^X, S^X) \text{ with } \partial S/\partial S^{\text{X}} > 0$$

$$Y = Y(K^Y, L^Y, S^Y) \text{ with } \partial Y/\partial S^{\text{Y}} > 0$$

$$S = S^X + S^Y$$

Both production activities involve emissions of S, and both individuals are adversely affected by the total amount of S emissions. In this case, efficiency requires that emissions from both sources are taxed at the same rate t = MEC*.

4.11 The second-best problem

In our discussion of market failure thus far we have assumed that just one of the ideal conditions required for markets to achieve efficiency is not satisfied. Comparing our list of the institutional arrangements required for markets to achieve efficiency with the characteristics of actual economies indicates that the latter typically depart from the former in several ways rather than just in one way. In discussing harmful externalities generated by firms, we have, for example, assumed that the firms concerned sell their outputs into perfectly competitive markets, are price-takers. In fact, very few of the industries in a modern economy are made up of firms that act as price-takers.

An important result in welfare economics is the second-best theorem. This demonstrates that if there are two or more sources of market failure, correcting just one of them as indicated by the analysis of it as if it were the only source of market failure will not necessarily improve matters in efficiency terms. It may make things worse. What is required is an analysis that takes account of multiple sources of market failure, and derives, as 'the second-best policy', a package of government interventions that do the best that can be done given that not all sources of market failure can be corrected.

To show what is involved, we consider in Figure 4.15 an extreme case of the imperfect competition problem mentioned above, that where the polluting firm is a monopolist. The analysis to be developed actually applies to any firm that faces a downward-sloping demand function for its output. It applies, that is, to all firms that get labelled 'imperfect competitors', or 'monopolistic competitors', in introductory microeconomics, as well as to monopolies strictly defined. It applies, then, to most firms in actual economies. However, in what follows here we will refer to the firm under consideration as 'the monopolist', as that usually reads more easily than 'the imperfectly competitive firm'.

As before, we assume that the pollution arises in the production of Y. The profit-maximising monopolist faces a downward-sloping demand function, D_YD_Y, and produces at the level where marginal cost equals marginal revenue, MR_Y. Given an uncorrected externality, the monopolist will use PMC here, and the corresponding output level will be Y_0. From the point of view of efficiency, there are two problems about the output level Y_0. It is too low on account of the monopolist setting marginal cost equal to marginal revenue rather than price – Y_c is the output level that goes with PMC = P_Y. It is too high on account of the monopolist ignoring the external costs generated and working with PMC rather than SMC – Y_t is the output level that goes with SMC = MR_Y. What efficiency requires is SMC = P_Y, with corresponding output level Y^*.

Now suppose that there is an EPA empowered to tax firms' emissions and that it does this so that for this monopolist producer of Y, SMC becomes the marginal cost on which it bases its decisions. As a result of the EPA action, Y output will go from Y_0 down to Y_t, with the price of Y increasing from P_{Y0} to P_{Yt}. The imposition of the tax gives rise to gains and losses. As intended, there is a gain in so far as pollution damage is reduced – the monetary value of this reduction is given by the area abcd in Figure 4.15. However, as a result of the price increase, there is a loss of consumers' surplus, given by the area P_{Yt}efP_{Y0}. It cannot be presumed generally that the gain will be larger than the loss. The outcome depends on the slopes and positions of PMC, SMC and D_YD_Y, and in any particular case the EPA would have to have all that information in order to figure out whether imposing the tax would involve a net gain or a net loss.

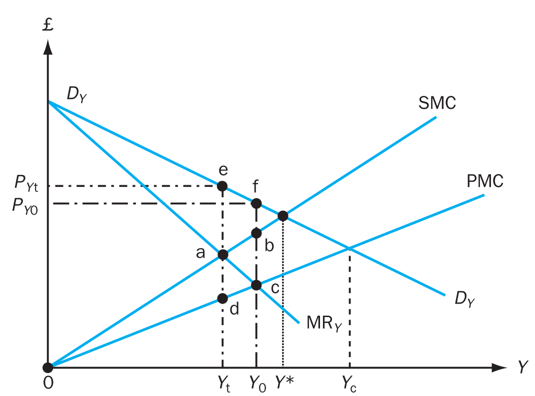

Figure 4.15 The polluting monopolist

When dealing with polluting firms that face downward-sloping demand functions, in order to secure efficiency in allocation the EPA needs two instruments – one to internalise the externality and another to correct under-production due to the firms setting MC = MR rather than MC = P. With two such instruments, the EPA could induce the firm to operate at Y^* where SMC = P_Y. However, EPAs are not given the kinds of powers that this would require. They can tax emissions, but they cannot regulate monopoly. As shown in Appendix 4.3.5, given complete information on the cost and demand functions, and on how damages vary with the firm's behaviour, the EPA could figure out a second-best tax rate to be levied on emissions. The second-best tax rate guarantees that the gains from its imposition will exceed the losses. The level of this second-best tax rate depends on the damage done by the pollutant, the firm's costs, and on the elasticity of demand for its output. With many polluting monopolies to deal with, the EPA would be looking at imposing different tax rates on each, even where all produce the same emissions, on account of the different elasticities of demand that they would face in their output markets. It needs to be noted that charging different firms different rates of tax on emissions of the same stuff is unlikely to be politically feasible, even if the EPA had the information required to calculate the different rates.

4.12 Imperfect information

Given that all of the other ideal institutional arrangements are in place, the attainment of efficient outcomes through unregulated market behaviour presupposes that all transactors are perfectly informed about the implications for themselves of any possible transaction. This is clearly a strong requirement, not always satisfied in actual market economies. The requirement carries over to the analysis of the correction of market failure. Consider, to illustrate the point here, a case of consumption-to-consumption external effect where two individuals share a flat (apartment) and where A is a smoker but B is not. Suppose that B does not find cigarette smoke unpleasant, and is unaware of the dangers of passive smoking. Then, notwithstanding that the government has legislated for property rights in domestic air unpolluted with cigarette smoke, B will not seek to reduce A's smoking. Given B's ignorance, the fact that bargaining is possible is irrelevant. The level of smoke that B endures will be higher than it would be if B were not ignorant. Given that B does not, when legally she could, bargain down A's level of smoking, we could describe the situation as one of 'conditional efficiency'. But this is not really very helpful. Rather, we recognise B's ignorance and consider it to be the source of an uncorrected externality. The nature of the corrective policy in the case of imperfect information is clear – the provision of information. In many cases, the information involved will have the characteristics of a public good, and there is a role for government in the provision of accurate information.

In some cases the government cannot fulfil this role because it does not have accurate and unambiguous information. Particularly where it is the future consequences of current actions that are at issue – as for example in the case of global warming – it may be simply impossible for anybody to have complete and accurate information. We all, as they say, live in an uncertain world. Imperfect information about the future consequences of current actions becomes particularly important in circumstances where those actions have irreversible consequences. It does appear to be the case that many of the consequences of decisions about environmental resource use are irreversible. Global warming may be a case in point. Again, it is arguable that once developed, a natural wilderness area cannot be returned to its natural state. We take up some of the issues arising from such considerations in Parts III and IV of the book.

4.13 Public choice theory – explaining government failure

We have shown that government intervention offers the possibility of realising efficiency gains, by correcting market failure. Many environmental resources are not subject to well-defined and clearly established property rights. In such circumstances, efficiency gains may be obtained if government can create and maintain appropriate institutional arrangements for establishing and supporting property rights as the basis for bargaining. The scope for this kind of government action to correct market failure is limited to cases where non-rivalry and non-excludability are absent. Many environmental problems do involve non-rivalry and non-excludability.

In such cases, possible government interventions to correct market failure are often classified into two groups. So-called command-and-control instruments take the form of rules and regulations prohibiting, limiting or requiring certain forms of behaviour. Monetary instruments – tax and subsidy systems, and marketable permits – are designed to create appropriate patterns of incentives for private behaviour. We have looked at taxation briefly in this chapter, and we shall explore all of these instruments in depth in Chapter 6. As noted immediately above, another form that government intervention to correct market failure could take is providing information, or funding research activity that can increase the stock of knowledge.

The arguments we have used so far in this chapter have all pointed to the possibility of efficiency gains arising from public-sector intervention in the economy. But actual government intervention does not always or necessarily realise such gains, and may entail losses. It would be wrong to conclude from an analysis of 'market failure' that all government intervention in the functioning of a market economy is either desirable or effective.

First, the removal of one cause of market failure does not necessarily result in a more efficient allocation of resources if there remain other sources of market failure. We discussed this in Section 4.11 above, using the case of the polluting monopolist as an illustration. A second consideration is that government intervention may itself induce economic inefficiency. Poorly designed tax and subsidy schemes, for example, may distort the allocation of resources in unintended ways. Any such distortions need to be offset against the intended efficiency gains when the worth of intervention is being assessed.

In some cases, the chosen policy instruments may simply fail to achieve desired outcomes. This is particularly likely in the case of instruments that take the form of quantity controls or direct regulation. An example, which we will deal with at length in

Chapter 17, is the use of quantity controls in fisheries policy. To address the free-access problem of over-exploitation, governments have, for example, adopted measures such as determining minimum mesh sizes for nets, setting a maximum number of days of permitted fishing, requiring a number of days in port for vessels. Such measures have generally met with very little success as fishermen have responded to the regulations by making behavioural adjustments to minimise their impact.

It is not the case that actual government interventions are always motivated by efficiency, or even equity, considerations. Adherents of the 'public choice' school of economics argue that the way government actually works in democracies can best be understood by applying to the political process the assumption of self-interested behaviour that economists use in analysing market processes. The analysis of why governments and legislatures adopt the economic policies that they do, and the consequences, is also sometimes known as the study of 'political economy'. A brief synopsis of some of the main general ideas is as follows.

Four classes of political agent are distinguished: voters, elected members of the legislature, workers in the bureaucracy, and pressure groups. Voters are assumed to vote for candidates they believe will serve their own interests. Legislators are assumed to maximise their chances of re-election. Bureaucrats are assumed to seek to enlarge the size of the bureaucracy, so improving their own career prospects. Pressure groups push special interests with politicians and bureaucrats. The argument is that given these motivations and circumstances, the outcome is not going to be a set of enacted policies that promote either efficiency or equity.

Politicians lack accurate information about voters' preferences. Voters lack reliable information about politicians' intentions. It is relatively easy for pressure groups to get their message across to politicians precisely because they focus on particular concerns arising from the strongly held views of a relatively small number of individuals or firms. Pressure groups access politicians directly, and via the bureaucracy. Bureaucrats, given their self-interest, amplify for politicians the messages from pressure groups that appear to call for a larger bureaucracy. They also control the flow of technical information to the politicians. The outcome of all this is, it is argued, an excessively large government doing, largely, things which keep, at least some, pressure groups happy, rather than things that reflect the preferences of the majority of voters. This is 'government failure'.

Box 4.5 introduces the ideas of one economist working in the public-choice area who takes issue with the assumptions about voters usually made there, though arriving at similarly pessimistic conclusions regarding policy outcomes.

Box 4.5 The myth of the rational voter

A key input to thinking in public choice theory is the observation that the way a vote is cast in an election is not, usually, decisive. With an electorate in the thousands, my most reasonable assumption is that my vote for my preferred candidate will make no difference to the chances of her getting elected, and, hence, my getting what I wanted. This is to be contrasted with my situation as a consumer, where my 'vote' is, usually, decisive. If I offer to pay the going price, I get what I want. This leads to the idea of 'rational ignorance'. If my vote does not count for much, it would be irrational to devote resources to finding out about the candidates and their policies on the issues. Again, in contrast, ignorance for a consumer is rarely rational – since my consumer 'vote' is decisive, it makes sense to find out about what I get with it.

There is, however, a fairly obvious problem with rational ignorance. If I proceed on the basis that my vote does not matter, why should I expend the effort involved in voting? Recall that a basic premise of public choice theory is self-interested behaviour. One answer might be, then, some form of altruism, a sense of civic responsibility. An alternative explanation, offered by Brennan and Lomasky (1993), is that of 'expressive voting'. The idea is that individuals derive pleasure from expressing themselves, it makes them feel good about themselves. Someone who votes green, the idea is, does so not solely because they want to, and hope to, see pro-environment policies adopted, but because doing so expresses their commitment to the environment. And, any personal costs, such as higher taxes, that might be associated

Box 4.5 ***continued***

with the expression are heavily discounted precisely because the probability that they will actually occur is so low.

In *The myth of the rational voter: why democracies choose bad policies* (Caplan, 2007), Caplan notes the need for an explanation for voting if voters are self-interested, and offers an explanation which is acknowledged to be similar to that of Brennan and Lomasky. Caplan's variant is 'rational irrationality'. According to it, people have irrational beliefs over which they have preferences. And, 'If agents care about both material wealth and irrational beliefs, then as the price of casting reason aside rises, agents consume less irrationally' (p. 123). Again, the point is that in regard to voting the perceived price of irrationality is low, whereas in regard to buying it is not. Thus, the perceived material cost of voting, say, for protectionist trade polices, which are definitely irrational according to Caplan, is low, whereas the perceived material cost of consuming on the basis of irrational belief can be very high – think of quack medicines. Consequently, the argument goes, individuals acting on the basis of irrational belief is common in voting, but rare in markets.

Caplan offers as evidence for this hypothesis the divergence between the beliefs of the general public and those of economists as revealed in a 1996 Survey of Americans and Economists on the Economy. This involved asking the same 37 questions about economics in interviews with 250 economics PhDs and 1510 randomly selected members of the public. Each question asked the respondent to give one of three responses, which were coded 0, 1 and 2. For example, after the prompt

> Regardless of how well you think the economy is doing, there are always some problems that keep it from being as good as it might be. I am going to read you a list of reasons some people have given for why the economy is not doing better than it is. For each one, please tell me if you think it is a major reason the economy is not doing better than it is, a minor reason, or not a reason at all.

A major reason got scored at 2, a minor reason at 1, and not a reason at 0. The following were the average scores on the 11 questions that went with this prompt:[10]

	Public	Economists
Taxes are too high	1.50	0.75
The federal deficit is too big	1.72	1.15
Foreign aid spending is too high	1.50	0.15
There are too many immigrants	1.25	0.22
Too many tax breaks for businesses	1.30	0.65
Education and job training are inadequate	1.55	1.61
Too many people are on welfare	1.61	0.72
Women and minorities get too many advantages under affirmative action	0.75	0.21
People place too little value on hard work	1.45	0.82
The government regulates business too much	1.25	0.95
People are not saving enough	1.38	1.50

The divergences between the economists' score and the public's here are pretty much replicated across the other questions asked in the survey

There is no doubt that the public and economists have, on average, different views on economic issues. Caplan's response to this, and other evidence to the same effect, is in two steps. First, that to the extent that there is divergence, it is because the public's answers are wrong. Second, that this is not properly seen as rational ignorance. In regard to the latter, he offers three strands of explanation. First, treating ignorance as rational ignores introspective evidence for commitment and belief. Second, that ignorance would not lead to a systematic divergence from the 'truth'. Third, that there is an alternative explanation in terms of rational irrationality.

Actually, it is not obvious that the survey results can be so straightforwardly interpreted as evidence for any kind of ignorance or irrationality. The problem is that for many of the questions it is not obvious that not agreeing on average with the, average, view of economists is simply a manifestation of error. Evaluations of economic performance necessarily involve subjective judgements and ethics. Economic

[10] The scores reported here were read off from graphical presentations in Caplan (2007).

Box 4.5 ***continued***

performance evaluation is not like evaluating the performance of car mechanics or dentists. Some people, for various reasons, care more about employment levels than inflation rates, or more about income distribution than its average level. Caplan takes it that freer trade is unequivocally a good thing so that favouring protectionist policies is either wrong or irrational. But what economic theory says is that, given certain assumptions, free trade means that the winners from it can compensate the losers and still be better off. There is no guarantee that compensation will be made. It is not necessarily ignorant or self-interestedly irrational for a voter who believes that she might be one of the losers to oppose a policy intended to make trade freer.

There is, however, no doubt that many individuals are ignorant about many of the matters that might be affected by their voting behaviour. These are not solely economic issues. This does raise problems for democracy. It is not difficult to point, as Caplan does, to examples of democratic societies adopting policies that appeal to 'public opinion' while being open to severe criticism on economic efficiency grounds. Caplan's response is the recommendation 'to rely more on private choice and the free market' (p. 197). But, as this chapter has shown, there are many situations, relating to the environment for example, where, even by the limited criteria of allocative efficiency, we cannot rely on 'the free market'.

Summary

Efficiency and optimality

Allocative efficiency exists when it is impossible to make one person better off except by making some other person(s) worse off. An efficient allocation may be unfair. An optimal allocation is an efficient allocation that satisfies the criterion of fairness embodied in the social welfare function.

Market equilibrium

If, and only if, certain stringent conditions are satisfied will the equilibrium in a system of markets be an efficient allocation.

Market failure

Actual economies do not satisfy the conditions of the ideal competitive economy. Agents do not have perfect information, markets are incomplete, markets are often not perfectly competitive, markets cannot supply public goods, and much of consumption and production behaviour generates external effects. These 'failures' will result in inefficient allocations of resources.

Public goods

Pure public goods non-rivalrous and non-excludable.

Externalities

An externality, or external effect, occurs when one agent's actions impact on another in a way that is unintended and uncompensated.

Environmental policy objectives

From the welfare economics perspective, there is a policy-relevant environmental problem when some environmental service/disservice is subject to market failure, so that the level of service/disservice is not that required for allocative efficiency. The policy objective is then to secure provision at the level required for allocative efficiency.

The second-best problem

If there are two or more sources of market failure, correcting just one of them will not necessarily improve matters with respect to allocative efficiency, and may make things worse.

Further reading

For a thorough general coverage of welfare economics principles, see Bator (1957), Baumol (1977), Just *et al.* (1982), Kreps (1990), Varian (1987) or Layard and Walters (1978), chapter 1. Cornes and Sandler (1996) is an excellent advanced treatment of the welfare economics of public goods and externalities. Baumol and Oates (1988) develop the theory of environmental economics, with special attention to policy, from the welfare economics of public goods and externalities: see also Dasgupta (1990), Fisher (1981), Johannson (1987), Mäler (1985) and McInerney (1976). Verhoef (1999) is a recent survey of externality theory in relation to environmental economics, and Proost (1999) surveys contributions from public-sector economics. Classic early articles on environmental externalities include Ayres and Kneese (1969) and D'Arge and Kogiku (1972). Our treatment of the second-best problem as it relates to an imperfectly competitive firm in Appendix A.4.3 draws on Barnett (1980); see also Baumol and Oates (1988).

The analysis of democratic governance in terms of self-interested behaviour by politicians, voters, bureaucrats and pressure groups was systematically developed by Buchanan: see, for example, Buchanan and Tullock (1980). Renner (1999) derives some implications for sustainability policy from the work of the 'Virginia school' associated with Buchanan. Brennan and Lomasky (1993) explore a model of voter behaviour close to that of Caplan (2007) discussed in Box 8.5. Everret in Dietz *et al.* (1993) considers the history of environmental legislation in the USA in the period 1970 to 1990 within a public-choice framework looking at the interactions between the classes of actors and events.

Discussion questions

1. 'If the market puts a lower value on trees as preserved resources than as sources of timber for construction, then those trees should be felled for timber.' Discuss.
2. Do you think that individuals typically have enough information for it to make sense to have their preferences determine environmental policy?
3. How is the level of provision of national defence services, a public good, actually determined? Suggest a practical method, that would satisfy an economist, for determining the level of provision.
4. Economists see pollution problems as examples of the class of adverse externality phenomena. An adverse externality is said to occur when the

decisions of one agent harm another in an unintended way, and when no compensation occurs. Does this mean that if a pollution source, such as a power station, compensates those affected by its emissions, then there is no pollution problem?

5. While some economists argue for the creation of private-property rights to protect the environment, many of those concerned for the environment find this approach abhorrent. What are the essential issues in this dispute?

Problems

1. Suppose that a wood pulp mill is situated on a bank of the River Tay. The private marginal cost (MC) of producing wood pulp (in £ per ton) is given by the function

   $$MC = 10 + 0.5Y$$

   where Y is tons of wood pulp produced. In addition to this private marginal cost, an external cost is incurred. Each ton of wood pulp produces pollutant flows into the river which cause damage valued at £10. This is an external cost, as it is borne by the wider community but not by the polluting firm itself. The marginal benefit (MB) to society of each ton of produced pulp, in £, is given by

   $$MB = 30 - 0.5Y$$

   a. Draw a diagram illustrating the marginal cost (MC), marginal benefit (MB), external marginal cost (EMC) and social marginal cost (SMC) functions.
   b. Find the profit-maximising output of wood pulp, assuming the seller can obtain marginal revenue equal to the marginal benefit to society derived from wood pulp.
   c. Find the pulp output which maximises social net benefits.
   d. Explain why the socially efficient output of wood pulp is lower than the private profit-maximising output level.
   e. How large would marginal external cost have to be in order for it to be socially desirable that no wood pulp is produced?

2. Demonstrate that equations 4.1 and 4.2 embody an assumption that there are no externalities in either consumption or production. Suppose that B's consumption of Y had a positive effect upon A's utility, and that the use of K by firm X adversely affects the output of firm Y. Show how the utility and production functions would need to be amended to take account of these effects.

3. In the chapter and in Appendix 4.3 we considered the two-person consumption-to-consumption externality. As invited in the Appendix, show that an efficient outcome could be realised if a planner required the sufferer to bribe the generator at the appropriate rate, and work out what that rate is.

4. In considering producer-to-consumer externalities in Appendix 4.3, it was stated that where there are multiple sources of emissions, and where only individuals suffer from pollution, each source should be taxed at the same rate. Prove this, and derive the tax rate.

5. Repeat Problem 4 for the case where pollution affects both lines of production as well as both individuals' utility.

PART II Environmental pollution

CHAPTER 5 Pollution control: targets

The use of coal was prohibited in London in 1273, and at least one person was put to death for this offense around 1300. Why did it take economists so long to recognize and analyze the problem?

Fisher (1981), p. 164

Learning objectives

At the end of this chapter, the reader should be able to

- understand the concept of a pollution target
- appreciate that many different criteria can be used to determine pollution targets
- understand that alternative policy objectives usually imply different pollution targets
- understand how in principle targets may be constructed using an economic efficiency criterion
- understand the difference between flow and stock pollutants
- analyse efficient levels of flow pollutants and stock pollutants
- appreciate the importance of the degree of mixing of a pollutant stock
- recognise and understand the role of spatial differentiation for emissions targets
- appreciate the meaning and significance of non-convexities for environmental policy
- appreciate the distinction between first-best and second-best modes of analysis, and use this to understand the debate about possible double dividends from tax reform packages

Introduction

In thinking about pollution policy, the economist is interested in two major questions:

1. How much pollution should there be?
2. Given that some target level of emissions (or pollution concentration) has been chosen, what is the best method of achieving that level?

This chapter deals in the main with the first of these questions; the second is addressed in the next chapter. Both questions are revisited in Chapter 7, where the analysis deals explicitly with the case of pollution policy where information is imperfect. We shall also revisit the issue of targets in Chapter 16, in which choices about emissions levels are linked explicitly to decisions about the way in which stocks of non-renewable resources (particularly fossil fuels) are extracted and used over time.

Our main task in this chapter, therefore, is to consider the question of how much pollution there should be or – to put things a different way – what should be the 'target' level of pollution. To introduce some key ideas, suppose that you work for a national environmental protection agency and have been given the task of producing a briefing note for policy makers on how much of some particular pollutant should be emitted annually at the national level. Let us say that the country is the USA, you work for the USEPA, and the pollutant in question is carbon dioxide emissions. Meanwhile, a colleague of yours has been given an identical task, but will be working independently of you. Why might you and your colleague arrive at different answers to this question?

Even if you are both rational individuals, are equally well informed, have access to the same analytical resources, and do not make mistakes in your calculations, there are several reasons why your answers may be different. Here are some of the principal reasons:

1. The results of your analysis will depend on the modelling framework that is employed. One important dimension of that framework is the breadth or extensiveness of the 'system' within which the question is being examined. If different modelling frameworks are used you are likely to arrive at different policy recommendations.
2. How much pollution there should be also depends on the goal that is being sought. You may disagree on what the goal is or should be, and so arrive at different answers.
3. You may interpret the question in different ways, and so arrive at different (or perhaps non-comparable) answers.

This chapter explores these and other related matters. We shall begin (in Section 5.1) by showing why one might expect to obtain different answers to the question of how much pollution there should be under alternative notions of the breadth of the relevant 'system'.

The bulk of this chapter (in Sections 5.4 through to 5.10) is concerned with the orthodox economic theory of pollution target setting, in which the objective is economic efficiency, the policy maker is operating under conditions of complete information, and the modelling framework is relatively narrow in senses to be explained as we go along. Later chapters will generalise matters in various ways.

The penultimate section looks at a complication that commonly arises in making environmental policy decisions. This complication arises from non-convexity in one or more of the relevant processes that determine the problem. Non-convexities have important implications for both targets and instruments, and typically generate uncertainties about behaviour of pollution mechanisms and outcomes (and so will also be considered further in the following two chapters).

Section 5.12 investigates the case for tax reform packages in which an environmental tax is introduced, the revenue from which is used to reduce the level of some other distortionary tax. The 'double dividend hypothesis' contends that such packages can not only lead to environmental improvements but can also do so at negative real cost. If so there is a double dividend: economic welfare is increased by improved environmental quality and is further increased as a result of removal of deadweight inefficiencies. If such a claim could be substantiated, it clearly would have major implications for environmental policy.

Finally, in Section 5.13 we return to the issue of objectives or goals of environmental policy, and examine how environmental policy targets might differ from those suggested by an economic efficiency criterion when other policy objectives are being sought.

5.1 Modelling frameworks

It was noted above that the target level of emissions (or pollution) that comes out from some process of analysis will depend on the modelling framework that is employed, and in particular the breadth or extensiveness of the system within which the question is being examined. This notion encompasses several related matters.

First of all, breadth might refer to geographical or political scope. In the thought experiment mentioned above, one analyst might choose to think of carbon dioxide emissions from US sources alone, independently of what is happening elsewhere. Put another way, she is assuming that CO_2 emissions from elsewhere are either irrelevant, or exogenous to the task at hand. The other may take a broader perspective: while still looking at matters from a 'what is best for the USA' perspective, he tries to take account of the endogeneity of other countries' emissions – actions in the USA are likely to influence actions taken by other countries; and actions of others will also have feedback effects on the welfare of US citizens. Yet another colleague might have chosen to investigate the problem in terms of what is best for the world as a whole, and in so doing infer what emissions should be allowable from US actors.

We shall address this particular set of questions in Chapter 9 in thinking about international environmental problems. At the moment, let us just say that intuition suggests that these three individuals are likely to arrive at different (probably very different) recommendations about targets for USA CO_2 emissions.

There is a second aspect of 'system extensiveness' that also matters. Is a partial equilibrium or a general equilibrium analysis being used? In practice, much of the work done by economists that deals with pollution policy has used a partial equilibrium perspective, in which pollution is viewed as a negative externality. A partial equilibrium approach looks at a single market or single activity (and its associated pollution) in isolation from the rest of the economic system in which the market or activity is embedded. This simplifies the problem, and may be justified if the phenomenon being studied is small in relationship to the system as a whole, and so the feedbacks and interactions between the market of interest and other markets are insignificant.[1]

Where interactions or feedbacks are not insignificant, however, it can be dangerous to use results from partial equilibrium analysis, and so one might instead choose to operate within a general equilibrium framework. Doing so puts results on a firmer, and more consistent, set of theoretical foundations. Indeed, many of the seminal works in environmental economics have adopted general equilibrium frameworks (see, for example, Baumol and Oates, 1988, and Cornes and Sandler, 1996), while continuing to see pollution problems in terms of externalities and missing markets. Again, intuition suggests that quantitatively different pollution targets will emerge from partial- and from general-equilibrium modelling frameworks. The role that energy efficiency improvements can play illustrates well this contrast between partial and general equilibrium analysis, as we show in Chapter 8.

A third aspect of the extensiveness of the system being studied follows from thinking more widely about what one might mean by a 'general' analysis. In other words, how does one specify the system in which pollution-generating activities are embedded? The development of environmental economics, and to a greater extent, ecological economics as distinct disciplines led some writers to take a comprehensive view of that system. This involved bringing the material and biological subsystems into the picture, and taking account of the constraints implied by economy–environment interactions.

A simple step in this direction came with incorporating natural resources into economic growth models. Then pollution can be associated with resource extraction and use, and 'best' levels of pollution emerge in the solution to the optimal growth problem. Pollution problems are thereby given a firmer material grounding and policies concerning pollution levels and natural resource uses are linked. Much of the work done in this area has been abstract, at a high level of aggregation, and is technically difficult. Nevertheless, we feel it is of sufficient importance to warrant study, and have devoted Chapter 16 to it.[2]

There have been more ambitious attempts to use the material balance principle (which was explained in Chapter 2) as a vehicle for investigating pollution problems. These try systematically to model interactions between the economy and the environment. Production and consumption activities draw upon materials and energy from the environment. Residuals from economic processes are returned to various environmental receptors (air, soils, biota and water systems). There may be significant delays in the timing of residual flows from and to the environment. In a growing economy, a significant part of the materials taken from the environment is assembled in long-lasting structures, such as roads, buildings and machines. Thus flows back to the natural environment may be substantially less than extraction from it over some interval of time. However, in the long run the materials balance principle points to equality between outflows and inflows. If we defined the environment broadly (to include human-made structures as well as the natural environment) the equality would hold perfectly at all times. While the masses of flows to and from the environment are identical, the return flows are in different physical

[1] There is a further matter here that we consider later, in Section 5.12. Partial equilibrium analysis can only give reliable guidance for policy when the rest of the economy is in full competitive equilibrium.

[2] Our reason for placing this material so late in the text is pedagogical. The treatment is technically difficult, and is best dealt with after first developing the relevant tools in Chapters 14 and 15.

forms and to different places from those of the original, extracted materials. A full development of this approach goes beyond what we are able to cover in this book, and so is not discussed further (beyond pointing you to some additional reading). Once again, though, what is clear is that recommendations about pollution targets are likely to be heavily influenced by the extent (if any) to which the boundaries of one's analysis encompass physical and biotic systems, as well as a more narrowly defined economic system.

5.2 Modelling pollution within an economic efficiency framework

In this section, we consider what the criterion of economic efficiency has to say about determining pollution targets. To do so, it will be instructive to begin by developing a framework for thinking about how pollution emissions and stocks are linked, and how these relate to any induced damage. The various relationships between economic activity, pollutant emission flows, the stocks of pollutants, and the damage that arises from those flows and stocks is represented schematically in Figure 5.1.

Economic activity generates emissions (or 'residual') flows that impose loads upon environmental systems. The extent to which these waste loads generate impacts that are associated with subsequent damage depends upon several things, including:

- the assimilative (or absorptive) capacity of the receptor environmental media;
- the existing loads on the receptor environmental media;
- the location of the environmental receptor media, and so the number of people living there and the characteristics of the affected ecosystems;
- tastes and preferences of affected people.

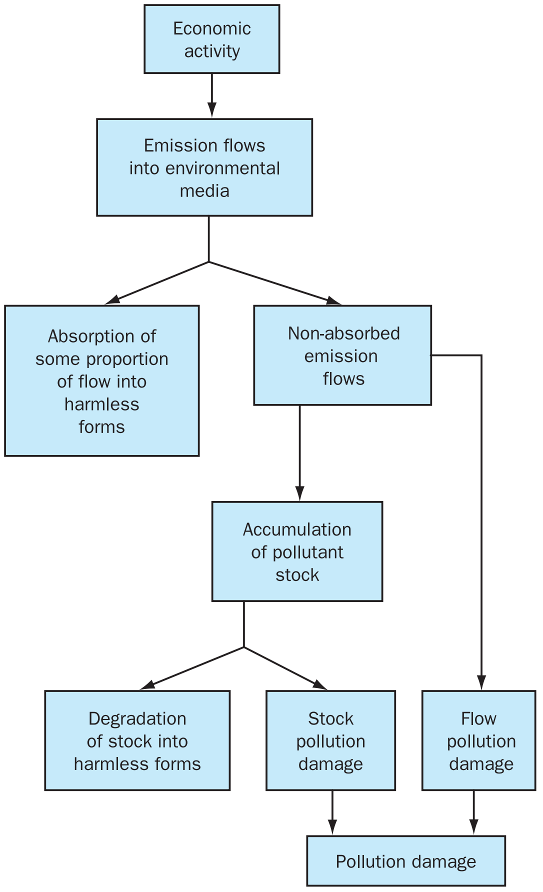

Figure 5.1 Economic activity, residual flows and environmental damage

Some proportion of the emission flows from economic activity is quickly absorbed and transformed by environmental media into harmless forms. The assimilative capacity of the environment will in many circumstances be sufficient to absorb and render harmless some amount of wastes. However, carrying capacities will often be insufficient to deal with all wastes in this way, and in extreme cases carrying capacities will become zero when burdens become excessive. Furthermore, physical and chemical processes take time to operate. Some greenhouse gases, for example, require decades to be fully absorbed in water systems or chemically changed into non-warming substances (see Table 5.1). This implies that some proportion of wastes will, in any time interval, remain unabsorbed or untransformed. These may cause damage at the time of their emission, and may also, by accumulating as pollutant stocks, cause additional future damage. Stocks of pollutants will usually decay into harmless forms but the rate of decay is often very slow. The half-lives of some radioactive substances are thousands of years, and for some highly persistent pollutants,

Table 5.1 Expected lifetimes for several pollutants

	Pre-industrial concentration	Concentration in 1998	Rate of concentration change	Atmospheric lifetime
CO_2 (carbon dioxide)	about 280 ppm	365 ppm	1.5 ppm/yr	5 to 200 yr[1]
CH_4 (methane)	about 700 ppb	1745 ppb	7.0 ppb/yr	12 yr
N_2O (nitrous oxide)	about 270 ppb	314 ppb	0.8 ppb/yr	114 yr
CFC-11 (chlorofluorocarbon-11)	Zero	268 ppt	1.4 ppt/yr	45 yr
HFC-23 (hydrofluorocarbon-23)	Zero	14 ppt	0.55 ppt/yr	260 yr
CF_4 (perfluoromethane)	40 ppt	80 ppt	1 ppt/yr	>50000 yr
Sulphur	Spatially variable	Spatially variable	Spatially variable	0.01 to 7 days
NO_X	Spatially variable	Spatially variable	Spatially variable	2 to 8 days

Note:
1. No single lifetime can be defined for CO_2 because of the different rates of uptake by different removal processes. Atmospheric lifetimes quoted here are taken directly from the source below, and are full estimated lifetimes, not half lives.
Sources: Technical Summary of the IPCC Working Group 1 Report, IPCC(1) (2001), Table 1, p. 38

such as the heavy metals, the rate of decay is approximately zero.[3]

5.3 Pollution flows, pollution stocks and pollution damage

Pollution can be classified according to whether damage arises from the flow of the pollutant (that is, the rate of emissions) or from the stock (or concentration rate) of pollution in the relevant environmental medium.[4] We define the following two classes of pollution: flow-damage pollution and stock-damage pollution (but recognise that there may also be mixed cases).

Flow-damage pollution occurs when damage results only from the flow of residuals: that is, the rate at which they are being discharged into the environmental system. This corresponds to the right-hand side branch in Figure 5.1. By definition, for pure cases of flow-damage pollution, the damage will instantaneously drop to zero if the emissions flow becomes zero. This can only be exactly true when the pollutant exists in an energy form such as noise or light so that when the energy emission is terminated no residuals remain in existence. However, this characterisation of damages may be approximately true in a wider variety of cases, particularly when the residuals have very short lifespans before being transformed into benign forms.

Stock-damage pollution describes the case in which damages depend only on the stock of the pollutant in the relevant environmental system at any point in time. This corresponds to the central branch in Figure 5.1. For a stock of the pollutant to accumulate, it is necessary that the residuals have a positive lifespan and that emissions are being produced at a rate which exceeds the assimilative capacity of the environment. An extreme case is that in which the assimilative capacity is zero, as seems to be approximately the case for some synthetic chemicals and a number of heavy metals. (The left-hand branch in Figure 5.1 does not then exist.) Metals such as mercury or lead accumulate in soils, aquifers and biological stocks, and subsequently in the human body, causing major damage to human health. Persistent synthetic chemicals, such as PCBs (polychlorinated biphenyls), DDT and dioxins, have similar cycles and effects. Rubbish which cannot biodegrade is another case. So are, for all practical purposes, strongly radioactive elements such as plutonium with extremely long radiation half-lives.

Most important pollution problems have the attribute of a stock-damage pollution effect being present. The most prominent are those which affect

[3] The half-life of a substance is the time it takes for a substance to lose half of its pharmacologic, physiologic, or radiologic activity.

[4] Box 5.1 gives some useful information regarding the distinction between stock and flow measures of a pollutant, illustrating the concepts with recent measures of carbon dioxide (CO_2) stocks and flows.

human health and life expectancy. But the phenomenon is more pervasive than this. Pollution stocks are harmful to built structures (buildings, works of art and so on) and they may adversely affect production potential, particularly in agriculture. Stock pollution levels influence plant and timber growth, and the size of marine animal populations. Less direct effects operate through damages to environmental resources and ecological systems. There is another way in which stock effects operate. The assimilative capacity of the environment often depends on the emissions load to which relevant environmental media are exposed. This is particularly true when the natural cleaning mechanism operates biologically. In water systems, for example, bacterial decomposition of pollutants is the principal cleaning agency. But where critical loads are exceeded, this biological conversion process breaks down, and the water system can effectively become dead. Its assimilative capacity has fallen to zero.

Mixed cases, where pollution damage arises from both flow and stock effects, also exist. Waste emissions into water systems are sometimes modelled as mixed stock-flow pollutants. So too are damages arising from the emissions of compounds of carbon, sulphur and nitrogen. However, in these mixed cases, it may often be preferable to view the problem as one of a pure stock pollutant.

Using M to denote the pollution flow, A to denote the pollution stock and D to denote pollution damage, we therefore have two variants of damage function:

$$\text{Flow-damage pollution: } D = D(M) \tag{5.1a}$$

$$\text{Stock-damage pollution: } D = D(A) \tag{5.1b}$$

For simplicity of notation, we shall from this point on call these 'flow pollution' and 'stock pollution'.

5.4 The efficient level of pollution

We now investigate how pollution targets can be set using an efficiency criterion. Given that pollution is harmful, some would argue that only a zero level of pollution is desirable. But, as we shall see, permitting some pollution can also be (indirectly) beneficial. Therefore, zero pollution is not economically efficient except in special circumstances.

In what sense is pollution beneficial? Clearly, pollution cannot be *intrinsically* beneficial; indeed, by definition, pollution refers to flows or stocks of

Box 5.1 Measures of pollutant stocks and flows

Pollutant emissions are measured (like all flows) in rates of output per period of time. For example, it is estimated that worldwide anthropogenic emissions of carbon dioxide, the most important greenhouse gas representing 77% of total anthropogenic greenhouse gas (GHG) emissions in 2004, were 21 gigatonnes of carbon dioxide per year (21 $GtCO_2$/yr) in 1970, rising to 38 gigatonnes in 2004. Taking all GHG together, with flows expressed in CO_2 equivalent rates, the growth of CO_2-eq emissions was much higher during the recent 10-year period of 1995–2004 (0.92 $GtCO_2$-eq per year) than during the previous period of 1970–1994 (0.43 $GtCO_2$-eq per year).

These flows accumulate through time as pollutant stocks, measured either in quantities in existence at some point in time, or in terms of some measure of concentration in an environmental medium of interest to us. Carbon dioxide atmospheric concentrations have risen from about 280 ppmv (parts per million by volume) in 1750 (the start of the industrial era) to 379 ppmv in 2005 (an increase of 35%). The current rate of change of the CO_2 concentration rate – measured over the 10-year period to 2005 – is estimated to be 1.9 ppmv per year (a growth rate of 0.5% per year).

Source: Climate Change 2007: Synthesis Report (IPCC). Available online at http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr.pdf (accessed 14 March 2009)

Note 1: A metric tonne is equal to 1000 kilograms (kg). Commonly used units for large masses are (i) a gigatonne (Gt) which is 10^9 tonnes, (ii) a megatonne (Mt) which is 10^6 tonnes, and (iii) a petragram (Pg) which is equal to 1 Gt. Finally, 1 GtC = 3.7 Gt carbon dioxide.

harmful residuals. So, other things being equal, less pollution is preferred to more. But it may not be possible to keep 'other things equal' as the level of pollution is altered. If producers of goods and services act rationally, they will select private cost-minimising techniques of production. Those techniques will often be ones that generate harmful emissions as joint products. Now suppose that a regulator requires that producers reduce the quantity of those harmful joint-products. In some circumstances – for example where production functions are Leontief in form (see Chapter 8) – emissions can only be reduced by decreasing the output of the marketed product. More generally, goods or services might be produced in less polluting ways, but at additional cost. Therefore, the imposition of stricter emissions standards is costly, in terms of lost output, the use of alternative inputs, or changed production techniques. It follows that if a required emissions standard is less restrictive, the relaxation of the pollution abatement constraint allows the production of goods that could not otherwise have been made, or to produce those goods at less direct cost in terms of inputs or techniques employed. This is the sense in which pollution could be described as (indirectly) beneficial.[5]

With both benefits and costs, economic decisions about the appropriate level of pollution involve the evaluation of a trade-off.[6] Stricter pollution targets will generate benefits but will also generate costs; the trade off-is optimised at the point where the marginal benefits arising from reduced pollution damage fall to a level equal to the marginal benefit from avoided control costs.

The discussion of efficient pollution targets which follows is divided into several parts. In Section 5.5 a static modelling framework is used to study efficient emissions of a flow pollutant. This explains the key principles involved in dealing with the trade-off. Sections 5.6 through to 5.10 investigate the more common – and important – case of stock-damage pollution. Two variants of stock damage are defined in Section 5.6. Sections 5.7 and 5.8 deal with those stock pollutants for which the location of the emission source matters as far as the pollutant stock, and so the extent of damages, is concerned. Our emphasis here will be on the *spatial* dimension of pollution problems. Section 5.9 focuses on the *time* dimension of pollution problems. It studies long-lived pollutants, such as greenhouse gases, which can accumulate over time. At this stage, our treatment of persistent stock pollutants will be relatively simple. Later, in Chapter 16, a richer dynamic modelling framework will be used to identify emission targets where pollution is modelled as arising from the depletion of natural resources.

5.5 A static model of efficient flow pollution

A simple static model – one in which time plays no role – can be used to identify the efficient level of a flow pollutant. In this model, emissions have both benefits and costs. In common with much of the pollution literature, the costs of emissions are called damages. Using a concept introduced in Chapter 4, these damages can be thought of as a negative (adverse) externality. Production entails joint products: the intended good or service, *and* the associated pollutant emissions. In an unregulated economic environment, the costs associated with production of the intended good or service are paid by the producer, and so are internalised. But the costs of pollution

[5] This argument is only legitimate if the rationality assumption is valid. If firms were not acting in a cost-minimising way and were able to make emissions cuts at no cost, then clearly one could not conclude that emissions reduction would be costly. Indeed, it is often argued that economies have many opportunities to make emissions reductions without incurring additional costs. We shall investigate this possibility a little further in the penultimate section of this chapter. For now, we proceed on the assumption that firms do make private cost-minimising choices. There is also an issue about whether the benefit function correctly describes the social benefits of emissions. Under some circumstances, emissions abatement can generate a so-called double dividend. If it does, the marginal benefit function as defined in this chapter will overstate the true value of emissions benefits. An extensive discussion of the double dividend idea is given later in Section 5.13.

[6] Our use of the term 'private' cost-minimising behaviour suggests that this can also be thought of as an externality problem, in which private costs diverge from social costs as a result of the presence of an external cost. Thinking about pollution as an externality arising from production or consumption activities makes this trade-off clear. The efficient level of an externality is not, in general, zero as the marginal costs of reducing the external effect will, beyond a certain point, exceed its marginal benefits.

damage are not met by the firm, are not taken into account in its decisions, and so are externalities. Moreover, in many cases of interest to us, it is also the case that the externality in question is what Chapter 4 called a public bad (as opposed to a private bad), in that once it has been generated, no one can be excluded from suffering its adverse effects. For simplicity, we suppose that damage is independent of the time and the source of the emissions, and that emissions have no effect outside the economy being studied. We shall relax these two assumptions later, the first in Section 5.7 and in Chapter 6, and the second in Chapter 9.

An efficient level of emissions is one that maximises the net benefits from pollution, where net benefits are defined as pollution benefits minus pollution damages. The level of emissions at which net benefits are maximised is equivalent to the outcome that would prevail if the pollution externality were fully internalised. Therefore, the identification of the efficient level of an adverse externality in Figure 4.14 (in Chapter 4), and the discussion surrounding it, is apposite in this case with an appropriate change of context.

In the case of flow pollution, damage (D) is dependent only on the magnitude of the emissions flow (M), so the damage function can be specified as

$$D = D(M) \tag{5.2}$$

Matters are less obvious with regard to the benefits of pollution. Expanding on the interpretation we offered earlier of the notion of pollution benefits, consider a representative firm. For any particular level of output it chooses to make, there will be an unconstrained emissions level that would arise from the cost-minimising method of production. If it were required to reduce emissions below that unconstrained level, and did so in the profit-maximising way, the total of production and control costs would exceed the total production costs in the unconstrained situation. So there are additional costs associated with emissions reduction. Equivalently, there are savings (or benefits) associated with emissions increases. It is these cost savings that we regard as the benefits of pollution. Symbolically, we can represent this relationship by the function

$$B = B(M) \tag{5.3}$$

in which B denotes the benefits from emissions.[7] The social net benefits (NB) from a given level of emissions are defined by

$$\text{NB} = B(M) - D(M) \tag{5.4}$$

It will be convenient to work with marginal, rather than total, functions. Thus $\mathrm{d}B/\mathrm{d}M$ (or $B'(M)$ in an alternative notation) is the marginal benefit of pollution and $\mathrm{d}D/\mathrm{d}M$ (or $D'(M)$) is the marginal damage of pollution. Analysis typically begins under the assumption that the total and marginal damage and benefit functions have the general forms shown in Figure 5.2. Total damage is thought to rise at an increasing rate with the size of the emission flow, and so the marginal damage will be increasing in M. In contrast, total benefits will rise at a decreasing rate as emissions increase (because per-unit emissions abatement costs will be more expensive at greater levels of emissions *reduction*). Therefore, the marginal benefit of emissions would fall as their flow increases.

It is important to understand that damage or benefit functions (or both) will not necessarily have these general shapes – shapes that are sometimes described as 'well behaved', or as satisfying convexity or concavity conditions. For many kinds of pollutants, the functions can have very different properties, as the discussion in Section 5.11 will illustrate. Where one or both of these functions are not well behaved, the results we obtain in this section may no longer hold. For now, except where it is stated otherwise, our presentation will assume that the general shapes shown in Figure 5.2 apply.

To maximise the net benefits of economic activity, we require that the pollution flow, M, be chosen so that

[7] Given our interpretation of the emissions benefit function (which involves optimised emissions abatement costs at any level of emissions below the unconstrained level), it will not be an easy matter to quantify this relationship numerically. However, there are various ways in which emissions abatement cost functions can be estimated. These are summarised in a document on the Companion Website entitled 'Estimating the cost of reducing pollution'. And, as we shall see later, with a suitable change of label abatement cost functions are identical to the benefit function we are referring to here.

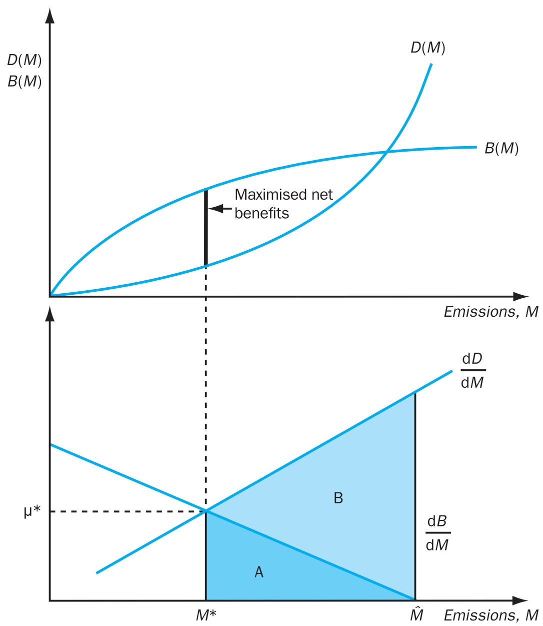

Figure 5.2 Total and marginal damage and benefit functions, and the efficient level of flow pollution emissions

$$\frac{\mathrm{dNB}(M)}{\mathrm{d}M} = \frac{\mathrm{d}B(M)}{\mathrm{d}M} - \frac{\mathrm{d}D(M)}{\mathrm{d}M} = 0 \tag{5.5a}$$

or, equivalently, that

$$\frac{\mathrm{d}B(M)}{\mathrm{d}M} = \frac{\mathrm{d}D(M)}{\mathrm{d}M} \tag{5.5b}$$

which states that the net benefits of pollution can be maximised only where the marginal benefits of pollution equal the marginal damage of pollution.[8] This is a special case of the efficiency condition for an externality stated in Chapter 4.

The efficient level of pollution is M^* (see Figure 5.2 again). If pollution is less than M^* the marginal benefits of pollution are greater than the marginal damage from pollution, so higher pollution will yield additional net benefits. Conversely, if pollution is greater than M^*, the marginal benefits of pollution are less than the marginal damage from pollution, so less pollution will yield more net benefits.

The value of marginal damage and marginal benefit functions at their intersection is labelled μ^* in Figure 5.2. We can think of this as the equilibrium 'price' of pollution. This price has a particular significance in terms of an efficient rate of emissions tax or subsidy, as we shall discover in the following chapter. However, as there is no market for pollution, μ^* is a hypothetical or shadow price rather than one which is actually revealed in market transactions. More specifically, a shadow price emerges as part of the solution to an optimisation problem (in this case the problem of choosing M to maximise net benefits). We could also describe μ^* as the shadow price of the pollution externality. If a market were, somehow or other, to exist for the pollutant itself (thereby internalising the externality) so that firms had to purchase rights to emit units of the pollutant, μ^* would be the efficient market price. Chapter 6 demonstrates that μ^* is the equilibrium price of tradable permits if an amount M^* of such permits were to be issued.

Another interpretation of the emissions efficiency condition (equation 5.5b) is obtained by inspection of Figure 5.3. The efficient level of pollution is the one that minimises the sum of total abatement costs plus total damage costs. Notice that in the diagram we have relabelled the curve previously called marginal benefit as marginal abatement cost. The logic here should be clear given our earlier discussion about the derivation of the benefits of pollution function.[9]

[8] This marginal equality applies when the optimum is at an interior point (does not fall at either extreme of the domain of the function). A sufficient second-order condition for this solution to be a local net benefit maximum is that $\mathrm{d}^2NB/\mathrm{d}M^2 = \mathrm{d}^2B/\mathrm{d}M^2 - \mathrm{d}^2D/\mathrm{d}M^2 < 0$. Both an interior solution and the second-order condition are satisfied given the slopes and relative positions of the functions assumed in the text and shown in Figure 5.2.

[9] The reinterpretation follows from the fact that reducing emissions incurs abatement costs. By construction, these (marginal) abatement costs are equal to the marginal benefits that will be lost if emissions fall. So, in Figure 5.3, if we start at the unconstrained emissions level, denoted as $\hat{M}$ in the diagram, then moving leftwards towards the origin corresponds to rising amounts of pollution abatement. Marginal abatement costs are low at small levels of abatement, and rise at an increasing rate as the abatement level becomes larger.

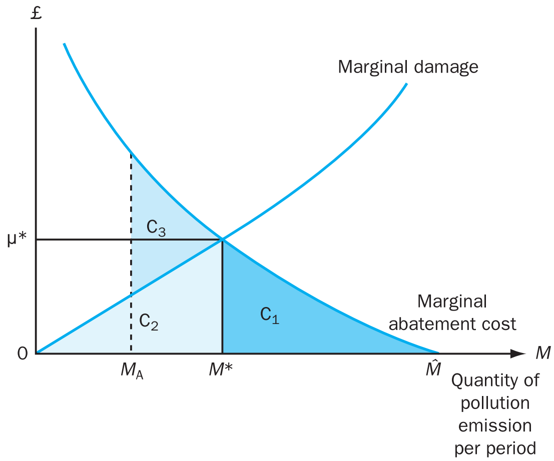

Figure 5.3 The economically efficient level of pollution minimises the sum of abatement and damage costs

To confirm this cost-minimising result, note that at the efficient pollution level, M^*, the sum of total damage costs (the area C_2) and total abatement costs (the area C_1) is $C_2 + C_1$. Any other level of emissions yields higher total costs. If too little pollution is produced (or too much abatement is undertaken) with a pollution flow restricted to M_A, it can be deduced that total costs rise to $C_1 + C_2 + C_3$, so C_3 is the efficiency loss arising from the excessive abatement. If you cannot see how this conclusion is reached, look now at Problem 2 at the end of this chapter. You should also convince yourself that too much pollution (too little abatement) results in higher costs than $C_1 + C_2$.

It can also be deduced from Figures 5.2 and 5.3 that the efficient level of pollution will not, in general, be zero. (By implication, the efficient level of pollution abatement will not, in general, correspond to complete elimination of pollution.) Problem 1 examines this matter.

We round off this section with a simple numerical example, given in Box 5.2. Functional forms used in the example are consistent with the general forms of marginal benefit and marginal damage functions shown in Figure 5.2. We solve for the values of M^*, B^*, D^* and μ^* for one set of parameter values. Also provided, in the *Additional Materials* that are linked to this text, is an Excel spreadsheet (*Targets*

Box 5.2 Efficient solution for a flow pollutant: a numerical example

Suppose that the total damage and total benefits functions have the following particular forms:

$$D = M^2 \text{ for } M \geq 0$$

$$B = \begin{cases} 96M - 0.2M^2 & \text{for} \quad 0 \leq M \leq 240 \\ 11\,520 & \text{for} \quad M > 240 \end{cases}$$

What is M^*?

If M is less than or equal to 240, then we have $B = 96M - 0.2M^2$ and so $\mathrm{d}B/\mathrm{d}M = 96 - 0.4M$. For any positive value of M we also have $D = M^2$ which implies that $\mathrm{d}D/\mathrm{d}M = 2M$. Now setting $\mathrm{d}B/\mathrm{d}M = \mathrm{d}D/\mathrm{d}M$ we obtain $96 - 0.4M = 2M$, implying that $M^* = 40$.

Substituting $M^* = 40$ into the benefit and damage functions gives us the result that $B^* = 3520$ and $D^* = 1600$, and so maximised total net benefits (NB*) are 1920. Note also that at M^* marginal benefit and marginal damage are equalised at 80 and so the shadow price μ^* – the value of value of marginal pollution damage at the efficient outcome – is 80.

You should now verify that $M^* = 40$ is a global optimum. This can be done by sketching the respective marginal functions and showing that net benefits are necessarily lower than 1920 for any (positive) level of M other than 40.

Additional materials

It can be useful to write a spreadsheet to do the kind of calculations we have just gone through. Moreover, if the spreadsheet is constructed appropriately, it can also serve as a template by means of which similar calculations can be quickly implemented as required. Alternatively, we could use such a spreadsheet to carry out comparative statics; that is, to see how the solution changes as parameter values are altered.

We have provided an Excel workbook *Targets examples.xls* that can be used in these ways in the *Additional Materials* available on the textbook's Web pages. That spreadsheet also shows how one of Excel's tools – 'Solver' – can be used to obtain the efficient level of M directly, by finding the level of M which maximises the net benefit function $\mathrm{NB} = B - D = (96M - 0.2M^2) - (M^2)$.

examples.xls) that reproduces these calculations. The Excel workbook is set up so that comparative statics analysis can be done easily by the reader. That is, the effects on M^*, B^*, D^* and μ^* of changes in parameter values from those used in Box 5.2 can be obtained.

5.6 Efficient levels of emission of stock pollutants

The analysis of pollution in Section 5.5 dealt with the case of flow pollution, in which pollution damage depends directly on the level of emissions. In doing so, there were two reasons why it was unnecessary to distinguish between flows and stocks of the pollutant. First, both benefits and damages depended on emissions alone, so as far as the objective of net benefit maximisation was concerned, stocks – even if they existed – were irrelevant. But we also argued that stocks do not exist for pure flow pollutants (such as noise or light).

How should the analysis change for stock pollutants, where damage depends on the stock of the pollutant? It turns out to be the case that the flow pollution model also provides correct answers in the special (but highly unlikely) case where the pollutant stock degrades into a harmless form more or less instantaneously. Here the stock dimension is distinguishable from the flow only by some constant of proportionality, and so we can work just as before entirely in flow units. But in all other cases of stock pollutants, the flow pollution model is invalid.

The majority of important pollution problems are associated with stock pollutants. Pollution stocks derive from the accumulation of emissions the life (or residence) time of which is more than merely instantaneous. The distinction between flows and stocks now becomes crucial for two reasons. First, without it, understanding of the science lying behind the pollution problem is impossible. Second, the distinction is important for policy purposes. While the damage is associated with the pollution stock, that stock is outside the direct control of policy makers. Environmental protection agencies may, however, be able to control the rate of emission flows. Even where they cannot control such flows directly, the regulator may find it more convenient to target emissions rather than stocks. Given that what we seek to achieve depends on stocks but what is controlled or regulated are typically flows, it is necessary to understand the linkage between the two.

Because stock pollutants are persistent over time and so may be transported over space, the analysis of stock pollution often necessitates taking account of space as well as time. It will be convenient to deal with these two dimensions separately. To do so, we draw a distinction between pollutants with a relatively short residence time (of the order of a day or so) and those with considerably longer lifetimes (years rather than days, let us say). Table 5.1, located near the start of this chapter, provides estimates of the active life expectancy of a range of pollutants under normal conditions.

5.7 Pollution control where damages depend on location of the emissions

In this section and the next we deal with stock pollutants which have relatively short residence times in the environmental media into which they are dumped. To help fix ideas, consider Figure 5.4 which represents two polluting 'sources', S_1 and S_2, that are located near four urban areas, R_1, R_2, R_3 and R_4. These areas contain populations whose health is adversely affected by local ambient concentrations of the pollutant. Our interest lies in the amount of pollution these areas – called 'receptors' – receive from the emission sources. We assume that emissions from the two sources persist for at most a few days; atmospheric processes break up and degrade concentrations rather quickly, so that on any one day pollutant concentrations are determined purely by emissions of the last few days. There is no long-term accumulation effect taking place.

Now consider the extent of pollutant dispersion and mixing. Mixing of a pollutant refers to the extent to which physical processes cause the pollutant to be dispersed or spread out. One possibility is that emissions are 'uniformly mixing' (UM). A pollutant is uniformly mixing if physical processes operate so that the pollutant quickly becomes dispersed to the

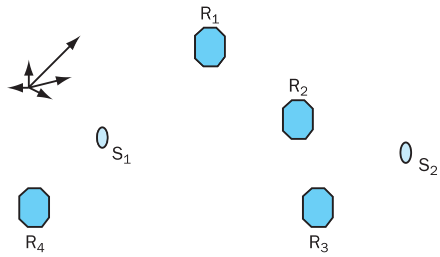

Figure 5.4 A spatially differentiated airshed

point where its spatial distribution is uniform. That is, the measured concentration rate of the pollutant does not vary from place to place. This property is satisfied, for example, by most greenhouse gases.

By definition, the location of the emission source of a UM pollutant is irrelevant as far as the spatial distribution of pollutant concentrations is concerned. Irrespective of the source location, pollutant stocks become evenly distributed across the whole spatial area of interest – in our picture over the whole rectangle depicted. All that matters, as far as concentration rates at any receptor are concerned, is the total amount of those emissions.

What can be said about the efficient level of emissions with the twin properties of short residence time (whose accumulation is therefore negligible) and uniform mixing? Intuition suggests that the simple flow pollution model developed in Section 5.5 can be used with only minor modification. To see why, note that there will be a one-to-one relationship between the level of emissions of the pollutant (M) and the pollutant stock size (A). Specifically, M and A are related by a fixed coefficient relationship of the form $A = kM$, with k fixed for any particular kind of pollution. Therefore, while damage is a function of the stock, and benefit is a function of flow, the damage function can be translated into an equivalent flow function using the $A = kM$ relationship, permitting use of the flow pollution model. A simple numerical example is given in Box 5.3. This has been reproduced as an Excel spreadsheet in Sheet 2 of the workbook *Targets examples.xls*. As was the case for the numerical example in Box 5.2, the Excel workbook has been set up to allow comparative static analysis to be carried out, and shows the use of Excel's Solver to obtain a direct solution to the optimisation problem.

The flow pollution model cannot be used where the pollutant is not uniformly mixing nor where it has a relatively long lifespan. (Can you explain why?) Most air, water and ground pollutants are not uniformly mixing. Look at Figure 5.4 again. Suppose that the principal determinants of the spatial distribution of the pollutant are wind direction and velocity.

Box 5.3 Efficient solution for a uniformly mixed and short-lived stock pollutant: a numerical example

We continue to use the notation M for emissions and A for concentrations. As in Box 5.2 we suppose that total benefits function is given by:

$$B = \begin{cases} 96M - 0.2M^2 & \text{for } 0 \le M \le 240 \\ 11\,520 & \text{for } M > 240 \end{cases}$$

Our total damage, however, now needs to be specified appropriately for a stock pollutant and is taken to be:

$$D = 0.2A^2 \text{ for } A \ge 0$$

and in steady state we assume that $A = 2M$.

What are M^* and A^*?

We first consider the case in which there is an interior solution with M positive but less than 240. The relevant first derivatives are:

$$\mathrm{d}B/\mathrm{d}M = 96 - 0.4M$$

$$\mathrm{d}D/\mathrm{d}M = 1.6M$$

(as $D = 0.2A^2$ implies $D = 0.2 \times (2M)^2 = 0.8M^2$ which implies $\mathrm{d}D/\mathrm{d}M = 1.6M$).

Now setting $\mathrm{d}B/\mathrm{d}M = \mathrm{d}D/\mathrm{d}M$ we obtain:

$$96 - 0.4M = 1.6M \rightarrow M^* = 48 \text{ and so } A^* = 96$$

Additional materials

A spreadsheet can be used for obtaining solutions to problems of this kind, or for carrying out comparative statics. Sheet 2 of the Excel workbook *Targets examples.xls* sets up a template for simple stock pollution models of this form. The interested reader may find it helpful to explore that sheet.

In the diagram, the length and direction of the arrow vectors in the multiple arrow symbol represent the relative frequency of these two components. Clearly, emissions from S_1 are going to matter more for the four receptor areas than emissions from S_2. Furthermore, looking at emissions from S_1 alone, these are likely to raise pollutant concentration levels to a greater amount in R_1 than in the other three receptors. R_4 is likely to suffer the least from emissions by either source.

Other factors will, of course, come into play. For example, suppose R_1 is at high elevation, whereas R_2 is situated in a depression surrounded by a ring of hills. Then R_2 may experience the highest concentrations, both on average and at peak times. All of this amounts to saying that where pollutants are not uniformly mixing, location matters. There will not be a single relationship between emissions and concentration over all space. A given total value of M will in general lead to differentiated values of A across receptors. Moreover, if M remained constant but its source distribution changed then the spatial configuration of A would also change.

Non-uniform mixing is of importance as many types of pollution fall into this category. Examples include ozone accumulation in the lower atmosphere, oxides of nitrogen and sulphur in urban airsheds, particulate pollutants from diesel engines and trace metal emissions. Many water and ground pollutants also do not uniformly mix. An environmental protection agency may attempt to handle these spatial issues by controlling *ex ante* the location of pollution creators and victims. This approach, implemented primarily by zoning and other forms of planning control, forms a substantial part of the longer-term way of dealing with spatial aspects of pollution. However, in the next section we focus on the situation in which the location of polluters and people is already determined, and moving either is not a feasible option. Our interest must then lie in how targets for emissions from the various sources can be calculated (and, in the next chapter, on what instruments can be used).

5.8 Ambient pollution standards

It will be convenient to use a little elementary matrix algebra for the exposition of the arguments that follow. For the reader unfamiliar with matrix algebra, or who needs a quick refresher, a brief appendix is provided for this purpose (Appendix 5.1, available on the Companion Website) explaining the notation used in matrix algebra and stating some simple results. It would be sensible to read that now.

Some additional notation is required. We regard the environment as a series of spatially distinct pollution ‘reception’ areas (or receptors). Suppose that there are J distinct receptors, each being indexed by the subscript j (so $j = 1, 2, \ldots, J$) and N distinct pollution sources, each being indexed by the subscript i (so $i = 1, 2, \ldots, N$). Various physical and chemical processes determine the impact on pollutant concentration in any particular receptor from any particular source. For simplicity, we assume that the relationships are linear. In that case, a set of constant ‘transfer coefficients’ can be defined. The transfer coefficient d_{ji} describes the impact on pollutant concentration at receptor j attributable to source i.[10] The total level, or concentration rate, of pollution at location j, A_j, will be the sum of the contributions to pollution at that location from all N emission sources. This can be written as

$$A_j = \sum_{i=1}^{N} d_{ji}M_i \tag{5.6}$$

where M_i denotes the total emissions from source i.

A numerical example will help. In the case shown in Figure 5.4, we have $N = 2$ sources and $J = 4$ receptors. Then we have four equations corresponding to equation 5.6. These are

$$A_1 = d_{11}M_1 + d_{12}M_2 \tag{5.7a}$$

$$A_2 = d_{21}M_1 + d_{22}M_2 \tag{5.7b}$$

$$A_3 = d_{31}M_1 + d_{32}M_2 \tag{5.7c}$$

$$A_4 = d_{41}M_1 + d_{42}M_2 \tag{5.7d}$$

[10] The linearity assumption is a very good approximation for most pollutants of interest. (As with some other secondary pollutants, low-level ozone accumulation is one significant exception.) Each coefficient d_{ji} will, in practice, vary over time, depending on such things as climate and wind conditions. However, if we measure average values of these coefficients over some period of time, they can be regarded as constant coefficients for the purposes of our analysis.

We can collect all eight d_{ji} coefficients into a $J \times N$ matrix, **D**. Denoting the vector of emissions from the two sources as **M** and the vector of ambient pollution levels in the four receptors as **A** we have

$$\mathbf{A} = \mathbf{DM} \tag{5.8}$$

or

$$\begin{bmatrix} A_1 \\ A_2 \\ A_3 \\ A_4 \end{bmatrix} = \begin{bmatrix} d_{11} & d_{12} \\ d_{21} & d_{22} \\ d_{31} & d_{32} \\ d_{41} & d_{42} \end{bmatrix} \begin{bmatrix} M_1 \\ M_2 \end{bmatrix} \tag{5.9}$$

Knowledge of the **M** vector and the **D** matrix allows us to calculate ambient pollution levels at each receptor. If, for example, **D** and **M** are

$$\mathbf{D} = \begin{bmatrix} 0.7 & 0.1 \\ 0.9 & 0.2 \\ 0.3 & 0.2 \\ 0.1 & 0.0 \end{bmatrix} \text{ and } \mathbf{M} = \begin{bmatrix} 10 \\ 20 \end{bmatrix}$$

then $A_1 = 9$, $A_2 = 13$, $A_3 = 7$ and $A_4 = 1$. The Excel workbook *Matrix.xls* and Word file *Matrix.doc* in the *Additional Materials* for Chapter 5 illustrate how this – and other similar – matrix calculations can be done using a spreadsheet program.

Armed with this terminology, we now answer the following question in a general way: what is the socially efficient level of emissions from each source? As in all previous cases in this chapter, it will be the set of emission levels that maximises net benefits. To see how this works here, note that there are N emission sources, and so our solution will consist of N values of M_i, one for each source. Benefits consist of the sum over all N sources of each firm's pollution benefits. So we have

$$B = \sum_{i=1}^{N} B_i(M_i)$$

Damages consist of the sum over all J receptor areas of the damage incurred in that area. That is,

$$D = \sum_{j=1}^{J} D_j(A_j)$$

Hence the net benefits function to be maximised (by appropriate choice of M_i, $i = 1, \ldots, N$) is

$$\text{NB} = \sum_{i=1}^{N} B_i(M_i) - \sum_{j=1}^{J} D_j(A_j) \tag{5.10}$$

By substitution of equation 5.6 into 5.10, the latter can be written as

$$\text{NB} = \sum_{i=1}^{N} B_i(M_i) - \sum_{j=1}^{J} D_j\left(\sum_{i=1}^{N} d_{ji} M_i\right) \tag{5.11}$$

A necessary condition for a maximum is that

$$\begin{aligned} \frac{\partial \text{NB}}{\partial M_i} &= B_i'(M_i) - \sum_{j=1}^{J} D_j'(A_j)\frac{\text{d}A_j}{\text{d}M_i} = 0 \quad \text{for } i = 1, \ldots, N \\ &= B_i'(M_i) - \sum_{j=1}^{J} D_j'(A_j) d_{ji} = 0 \text{ for } i = 1, \ldots, N \end{aligned} \tag{5.12}$$

which, after rearranging, yields the set of N marginal conditions

$$B_i'(M_i) = \sum_{j=1}^{J} D_j'(A_j) d_{ji} \text{ for } i = 1, \ldots, N \tag{5.13}$$

where

$$D_j'(A_j) = \frac{\partial D_j}{\partial A_j}$$

The intuition behind this result is straightforward. The emissions target (or standard) for each firm should be set so that the private marginal benefit of its emissions (the left-hand side of the equation) is equal to the marginal damage of its emissions (the right-hand side of the equation). Note that because the ith firm's emissions are transferred to some or all of the receptors, the marginal damage attributable to the ith firm is obtained by summing its contribution to damage over each of the J receptors.

An interesting property of the solution to equation set 5.13 is that not only will the efficient emission level differ from firm to firm, but also the efficient ambient pollution level will differ among receptors. It is easy to see why efficient emission levels should vary. Firms located at different sources have different pollution impacts: other things being equal, those sources with the highest pollution impact should emit the least. But what lies behind the result that efficient levels of pollution will vary from place to place? Receptors at different spatial locations will experience different pollution levels: other things being equal (including population densities), those receptors which would in an unconstrained world experience the highest pollution-stock level should have the highest efficient ambient pollution level. Of course, these two considerations have to be met jointly; NB = $B - D$ is being maximised, and so we are searching for the best trade-off between the benefits reduction and damages reduction. Appendix 5.2 provides a worked numerical example of efficient emissions that illustrates this point.

In practice, environmental regulators might deem that it is unethical for A to vary from place to place. So, they might impose an additional constraint on the problem to reflect this ethical position. One form of constraint is that the pollution level in no area should exceed some maximum level A^* (that is $A_j^* \le A^*$ for all j). Another, stricter, version would be the requirement that A should be the same over all areas (that is $A_j^* = A^*$ for all j). In the latter case, the net benefit function to be maximised is

$$L = \sum_{i=1}^{N} B_i(M_i) - \sum_{j=1}^{J} D_j(A^*) \tag{5.14}$$

By imposing additional constraints, maximised net benefit is lower in equation 5.14 than in equation 5.10. An efficiency loss has been made in return for achieving an equity goal.

5.9 Intertemporal analysis of stock pollution

We now consider the case of stock pollutants that have a relatively long active (i.e. damaging) lifespan but which are uniformly mixing. Doing so has two implications. First, the uniformly mixing assumption implies that pollutant concentrations will not differ from place to place, and so the spatial dimension of emissions control is no longer of direct relevance. Second, persistence of pollution stocks over time means that the temporal dimension is of central importance. As we shall see, an efficient pollution control programme will need to take account of the trajectory of emissions over time, rather than just at a single point in time.

The model we use to examine pollution targets is the simplest possible one that can deal with the intertemporal choices involved. Damage at time t is determined by the contemporaneous stock size or concentration of the pollutant in a relevant environmental medium. Gross benefits depend on the flow of emissions. Hence our damage and (gross) benefit functions have the general forms

$$D_t = D(A_t) \tag{5.15}$$

$$B_t = B(M_t) \tag{5.16}$$

The variables A and M in equations 5.15 and 5.16 are, of course, not independent of one another. With relatively long-lived pollutants, emissions add to existing stocks and those stocks accumulate over time. However, except in the special case where pollutants are infinitely long-lived, part of the existing stock will decay or degrade into a harmless form over time, thereby having a negative impact on stock accumulation. A convenient way of representing this stock–flow relationship is by assuming that the rate of change of the pollutant stock over time is governed by the differential equation

$$\dot{A}_t = M_t - \alpha A_t \tag{5.17}$$

where a dot over a variable indicates its derivative with respect to time, so that $\dot{A}_t = \mathrm{d}A/\mathrm{d}t$. To interpret this equation, it will be helpful to have an example in mind. Consider atmospheric carbon dioxide (CO_2), one source of which is emissions from the combustion of fossil fuels. Current emissions (M_t) add to CO_2 stocks, and so the concentration level rises; that is, $\dot{A}_t$ is positive. However, offsetting factors are at work too. Some of the existing CO_2 stock will be transformed into harmless substances by physical or chemical processes, or will be absorbed into oceans or other sinks where it has no damaging effect. In other words, part of the pollution stock decays. The amount of pollution decay is captured by the term αA_t.

The net effect on A (and so whether $\dot{A}_t$ is positive or negative overall) depends on the magnitudes of the two terms on the right-hand side of equation 5.17.[11] The parameter α is a proportion that must lie in the interval zero to one. A pollutant for which $\alpha = 0$ exhibits no decay, and so the second term on the right-hand side of equation 5.17 is zero. This is known as a perfectly persistent pollutant. In this special case, integration of equation 5.17 shows that the stock at any time is the sum of all previous emissions. Notice that the absence of decay means that damages arising from current emissions will last indefinitely. This is approximately true for some synthetic chemicals, such as heavy metal residuals, and toxins such as DDT and dioxin. Moreover, the pollution stock and pollution damages will increase without bounds through time as long as M is positive.

More generally, we expect to find $0 < \alpha < 1$, and denote this as an imperfectly persistent pollutant. Here, the pollutant stock decays gradually over time, being converted into relatively harmless elements or compounds. Greenhouse gases provide one example, but (as we show in Chapter 9) with slow or very slow rates of decay. The second limiting case, where $\alpha = 1$, implies instantaneous decay, and so the pollutant can be regarded as a flow rather than a stock pollutant. We need deal with this special case no further here.

The specification given in equation 5.17 imposes the restriction that the parameter α is constant; a constant *proportion* of the pollution stock decays over any given interval of time. This may be invalid in practice. If the restriction is approximately true equation 5.17 might still be used for reasons of convenience and simplicity. But if it is grossly inaccurate, and the decay rate (or assimilation rate as it is often called) changes substantially over time, or varies with changes in either A or M, then it is not an appropriate basis for modelling. We will return to this matter later.

We mentioned earlier that, unlike in the previous cases investigated in this chapter, the relationship between M and A is not independent of time. By integrating equation 5.17 over time we obtain

$$A_t = \int_{\tau=t_0}^{\tau=t} \left(M_t - \alpha A_t\right) \mathrm{d}\tau$$

where t_0 denotes the first point in time at which the pollutant in question was emitted. Thus the pollution stock level at any time t, A_t, depends on the entire history of emissions up to that point in time. Even if emissions had been at a constant level in the past and were to remain so in the future, A would not be constant throughout time, except in the very long run. Put another way, as emissions at time t add to pollution stocks at that time *and* in future time periods, there is no one-to-one relationship between A and M. It is because time matters here in a fundamental way that the variables in equations 5.15 and 5.16 are time-dated.[12]

As time periods are linked together through a stock–flow relationship, efficient pollution targets and policies must be derived from an intertemporal analysis. We proceed by assuming that the policy maker aims to maximise discounted net benefits over some suitable time horizon. For simplicity, the horizon is taken to be of infinite span. Using $t = 0$ to denote the current period of time, and defining the net benefits of pollution as gross benefits minus damages (specified respectively by equations 5.16 and 5.15), the policy maker's objective is to select M_t for $t = 0$ to $t = \infty$ to maximise

[11] In this chapter, we are working principally with economic models specified in continuous-time terms. However, sometimes it is convenient to work in a discrete-time framework. Doing this requires defining the meaning to be attached to time subscripts for stock variables. A convention that we follow throughout this text is that for any stock variable the subscript t denotes the end of period t. Then the discrete-time counterpart of equation 5.17 would be:

$$A_t - A_{t-1} = M_t - \alpha A_{t-1}$$

Notice that the last term on the right-hand side now has the time subscript $t - 1$, as compared with t in equation 5.17. Given our convention, A_{t-1} refers to the pollution stock at the end of period $t - 1$ (or, equivalently, start of period t). The discrete-time counterpart of equation 5.17 would then say that the inflow (new emissions) is taking place contemporaneously with the outflow (stock decay), and that it is the difference between inflow and outflow during period t that determines whether stock will rise, fall or remain constant between the end of period $t - 1$ and the end of period t. This is intuitively sensible.

[12] In the last section, the relationship between stocks and flows of the pollutant was complicated because space mattered; the effect of M on A depended on the respective locations of the pollution source and recipient. There we used i and j terminology to denote that dependence on location. Here the relationship is complicated by the fact that time matters, hence the use of t terminology.

$$\int_{t=0}^{t=\infty} (B(M_t) - D(A_t))e^{-rt}dt \tag{5.18}$$

where r is the social (consumption) discount rate.

A complete description of efficient stock pollution will, therefore, consist not of a single number for, but a *trajectory* (or time path) of, emission levels through time. In general, this optimal trajectory will be one in which emission levels vary throughout time. However, in many circumstances, the trajectory will consist of two phases. One of these phases is a so-called *steady state* in which emissions (and concentration levels) remain constant indefinitely at some level. The other is an adjustment phase; the trajectory describes a path by which emissions (and concentrations) move from current levels to their efficient, steady-state levels. This adjustment process may be quick, or it may take place over a long period of time.

Even with complete information, obtaining such a trajectory is technically difficult, involving the calculus of optimal control. We will explain this technique in Chapter 14, and apply it to the pollution model being examined here in Chapter 16. In this chapter, we consider only the second of the two phases described above: the efficient *steady-state* pollution level.[13] In a steady state, by definition, the pollution flow and the pollution stock are each at a constant, unchanging level.[14] Hence the time subscripts we have attached to variables become redundant and can be dropped. Moreover, with an unchanging stock $\dot{A}_t = 0$ and so equation 5.17 simplifies to $M = \alpha A$. The intuition that lies behind this is straightforward: for a pollutant that accumulates over time, the pollution stock can only be constant if emission inflows to the stock (M) are equal to the amount of stock which decays each period (αA). It then follows that in a steady state, the stock–flow relationship between A and M can be written as

$$A = \frac{M}{\alpha} \tag{5.19}$$

This shows that, in a steady state, the smaller is the value of α the larger will be the pollution stock for any given level of emissions.

The full derivation of the steady-state solution to this problem is presented in Chapter 16. You may wish to return to, and reread, this section after studying that later chapter. Here, we just state one major result from that solution, interpret it intuitively, and discuss some of its characteristics. The authors ask you to take this result on trust; little is lost by not going through its derivation at this stage.

The key result we draw upon from Chapter 16 is that an efficient steady-state level of pollution emissions requires that the following condition be satisfied:

$$\frac{dB}{dM} = \frac{dD}{dA}\left(\frac{1}{r + \alpha}\right) \tag{5.20}$$

Equation 5.20 is a variant of the familiar marginal condition for efficiency. The marginal benefit and the marginal cost of the chosen emissions level should be equal. More precisely, it can be read as an equality between the instantaneous value of the gross benefit of a marginal unit of pollution (the left-hand side of 5.20) and the present value of the damage that arises from the marginal unit of pollution (the right-hand side of 5.20).[15] The discount factor $1/(r + \alpha)$ has the effect of transforming the single period damage into its present-value equivalent. (A fuller explanation of this interpretation is given in Chapter 16.) At the level of M that satisfies equation 5.20, the value taken by the expression on each side of the equation is known as the shadow price of a unit of emission. It is labelled as μ in several of the diagrams in this chapter and will figure prominently in our discussions in the next chapter.[16]

[13] Doing this assumes that the problem is one in which a steady-state solution exists, which is not always true. Chapter 16 will briefly examine the adjustment process to a steady state, and whether such a state exists.

[14] There is a second sense in which the term steady state is sometimes used: as a state in which all variables of interest in some system are growing at a constant rate. We do not use this alternative meaning in this text.

[15] Note that while the damage arising from the marginal emission takes place today and in future periods, a marginal emission today has benefits only today, and so the instantaneous value of the marginal benefit is identical to its present value. So we could also interpret equation 5.20 in terms of an equality between two present values.

[16] In some of the economics literature, the shadow price of emissions is constructed to be a negative quantity (and would correspond here to the negative of μ). This arises because some authors choose to attach a different interpretation to the shadow price. Whenever a different interpretation is being used in this text, it will be made clear to the reader explicitly.

Examination of equation 5.20 shows two very important results:

1. Other things being equal, the faster is the decay rate, the higher will be the efficient level of steady-state emissions. Reasoning: For any given value of dD/dA, a rise in α implies that the value of dB/dM would have to fall to satisfy the marginal equality. A lower value of dB/dM implies higher emissions. Intuition: The greater is the rate of decay the larger is the 'effective' discount rate applied to the marginal stock damage term and so the smaller is its present value. A higher discount rate means we attach less weight to damages in the future, and so the emission level can be raised accordingly.
2. Other things being equal, the larger is the consumption discount rate, the higher will be the efficient level of steady-state emissions. Reasoning: For any given value of dD/dA, a rise in r implies that the value of dB/dM would have to fall to satisfy the marginal equality. A lower value of dB/dM implies higher emissions. Intuition: The greater is the consumption discount rate r, the larger is the discount rate applied to the stock damage term and so the smaller is its present value. A higher discount rate means we attach less weight to damages in the future, and so the emission level can be raised accordingly.

Problem 4 at the end of this chapter asks the reader to explore these and other results from the stock pollution model. The model is simulated in the Excel workbook *Stock1.xls*.

For the purpose of looking at some special cases of equation 5.20, it will be convenient to use an alternative version of that expression as follows (the full derivation is given in Chapter 16):

$$\frac{\mathrm{d}D}{\mathrm{d}M} = \frac{\mathrm{d}B}{\mathrm{d}M}\left(1 + \frac{r}{\alpha}\right) \tag{5.21}$$

Four special cases of equation 5.21 can be obtained, depending on whether $r = 0$ or $r > 0$, and on whether $\alpha = 0$ or $r > 0$. We portray these combinations in Table 5.2.

Table 5.2 Special cases of equation 5.21

	Imperfectly persistent pollutant $\alpha > 0$	Perfectly persistent pollutant $\alpha = 0$
$r = 0$	A	D
$r > 0$	B	C

Case A: $r = 0$, $\alpha > 0$

In this case the pollutant is imperfectly persistent and so eventually decays to a harmless form. With $r = 0$, no discounting of costs and benefits is being undertaken. Equation 5.21 collapses to:[17]

$$\frac{\mathrm{d}D}{\mathrm{d}M} = \frac{\mathrm{d}B}{\mathrm{d}M} \tag{5.22}$$

This has a straightforward interpretation. An efficient steady-state rate of emissions for a stock pollutant requires that the contribution to benefits from a marginal unit of pollution flow be equal to the contribution to damage from a marginal unit of pollution flow. The steady-state equilibrium is shown in Figure 5.5 (by the intersection of the functions dD/dM and dB/dM). Net benefits are maximised at the steady-state pollution flow M*. In the steady state, A* will be at the level at which $\alpha A^* = M^*$, and both the pollution stock and emissions track along through time at constant levels. You may find it useful to look at Box 5.4 at this point; this goes through a simple numerical example to illustrate the nature of the equilibrium.

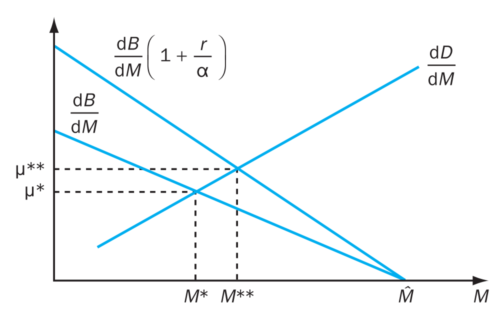

Figure 5.5 Efficient steady-state emission level for an imperfectly persistent stock pollutant. Two cases: {$r = 0$ and $\alpha > 0$} and {$r > 0$ and $\alpha > 0$}

[17] Notice that equation 5.22 appears to be identical to the efficiency condition for a flow pollutant. But it is necessary to be careful here, as 5.22 holds only in a steady state, and is not valid outside those states for a stock pollutant.

Box 5.4 Steady-state efficient solution for a stock pollutant: a numerical example

No discounting, $r = 0$ (Case A: $r = 0$, $\alpha > 0$)

Let $\alpha = 0.5$, $D = A^2$, $B = 96M - 2M^2$.
What are M^* and A^*?

$$B = 96M - 2M^2 \rightarrow \mathrm{d}B/\mathrm{d}M = 96 - 4M$$

$$D = A^2 = (M/\alpha)^2 = (1/0.5)^2M^2 = 4M^2 \rightarrow \mathrm{d}D/\mathrm{d}M = 8M$$

Now setting $\mathrm{d}B/\mathrm{d}M = \mathrm{d}D/\mathrm{d}M$ we obtain:

$$96 - 4M = 8M \rightarrow M^* = 8$$

Therefore $A = (M/\alpha) \rightarrow A^* = 16$

This result is obtained by inspection and by use of Solver in Sheet 1, and shown graphically in Chart 1, of Excel workbook *Stock1.xls* in the *Additional Materials* for Chapter 5.

Positive discounting, $r > 0$ (Case B: $r > 0$, $\alpha > 0$)

Let $\alpha = 0.5$, $r = 0.1$, $D = A^2$, $B = 96M - 2M^2$.
What are M^* and A^*?

$$B = 96M - 2M^2 \Rightarrow \mathrm{d}B/\mathrm{d}M = 96 - 4M$$

$$D = A^2 = (M/\alpha)^2 = (1/0.5)^2M^2 = 4M^2 \rightarrow \mathrm{d}D/\mathrm{d}M = 8M$$

Now setting $\frac{\mathrm{d}D}{\mathrm{d}M} = \frac{\mathrm{d}B}{\mathrm{d}M}\left(1 + \frac{r}{\alpha}\right)$ we obtain:

$$8M = (96 - 4M)(1 + \{0.1/0.5\}) \rightarrow M^* = 9$$

Therefore $A = (M/\alpha) \rightarrow A^* = 18$

This result is obtained by inspection and by use of Solver in Sheet 2, and shown graphically in Chart 2, of Excel workbook *Stock1.xls*. Note that we use Solver there to find the value of M that sets marginal net benefits (expressed in terms of emissions) equal to zero.

Case B: $r > 0$, $\alpha > 0$

With r and α being positive numbers, the equilibrium condition is given by equation 5.21 in unchanged form. The marginal equality in this case incorporates the additional term $1/(r + \alpha)$ to reflect the presence of discounting at a positive rate. This is shown diagrammatically in Figure 5.5, with M^{**} denoting the equilibrium emission level. It is instructive to compare this equilibrium with that obtained in Case A. As r increases above zero, the marginal damages function rotates clockwise about the point $\hat{M}$. Discounting, therefore, increases the steady-state level of emissions. Moreover, the larger is the discount rate, the larger is the amount by which efficient steady-state emissions rise. Intuitively, a larger value of r reduces the present value of the future damages that are associated with the pollutant stock. In effect, higher weighting is given to present benefits relative to future costs the larger is r. However, notice that the shadow price of one unit of the pollutant emissions becomes larger as r increases.

Cases C ($r > 0$, $\alpha = 0$) and D ($r = 0$, $\alpha = 0$)

In both Cases C and D the pollutant is perfectly persistent, and so never decays to a harmless form. One might guess that something unusual is happening here by noting that equation 5.21 is undefined when $\alpha = 0$; division by zero is not a legitimate mathematical operation. The intuition that lies behind this is straightforward. No steady state exists except for the case in which M is zero. A steady state cannot exist for any positive value of M as A would rise without bound. But then pollution damage would also rise to infinity.

It follows that, at some point in time, the environmental protection agency (EPA) would have to require that emissions be permanently set to zero to avoid the prospect of intolerable damage. The pollution stock level would remain at whatever level A had risen to by that time. Pollution damage would also continue indefinitely at some constant level, but no additional damage would be generated. The zero-emissions steady-state solution turns out to be perfectly in accord with good sense.

One caveat to this conclusion should be noted. Although a perfectly persistent pollutant has a zero natural decay rate, policy makers may be able to find some technique by which the pollutant stock may be artificially reduced. This is known as clean-up expenditure. If such a method can be found, and can be implemented at reasonable cost, it allows the possibility of some perpetual level of emissions. We examine this possibility further in Chapter 16.

Of course, even if the EPA accepted that emissions would have to be set to zero at some date (and remain zero thereafter), the question remains of which date the switch to zero should be made. Steady-state analysis is unable to answer this question. To obtain that answer, another technique (or another criterion than economic efficiency) is required. Chapter 16 shows how optimal control can be used to find both the efficient steady-state solution and the optimal adjustment path to it.

5.10 Variable decay

The stock pollution models used in this chapter have assumed that the proportionate rate of natural decay of the stock, α, is constant. This assumption is commonly employed in environmental economics analysis, but will not always be valid. In many situations, one would expect that the rate of decay depends on the size of the pollution stock, or on some other associated variable. For example, it is thought that the decay rate of greenhouse gases alters (in quite complex ways) as mean temperature levels change. Of particular importance are the existence of threshold effects (where the decay rate changes in a sudden, discontinuous way) and irreversibilities (where the nature of a relationship changes depending on the direction in which a variable is moving). One example of a threshold effect is illustrated in the top panel of Figure 5.6. Here the decay rate of a water-borne pollutant collapses towards zero as some threshold level of biological oxygen demand (BOD) on a river is reached. This critical level of BOD is reached when the pollution stock is at *A*. The lower panel illustrates a threshold effect combined with an irreversibility. The arrows denote the direction in which *A* is changing. As the pollution stock rises from a low level, α collapses suddenly at the threshold *A* and remains close to zero as *A* continues to rise. At high levels of pollution, the biological ability of the river to break down harmful pollutants might be largely destroyed. So if the change is reversed, and *A* falls from a high value, the value of α would remain very low (as shown by the path with left-pointing arrows). This path dependence is also known as hysteresis; in this example, the history of pollutant flows matters, and reversing pollution pressures does not bring one back to the *status quo ex ante*.

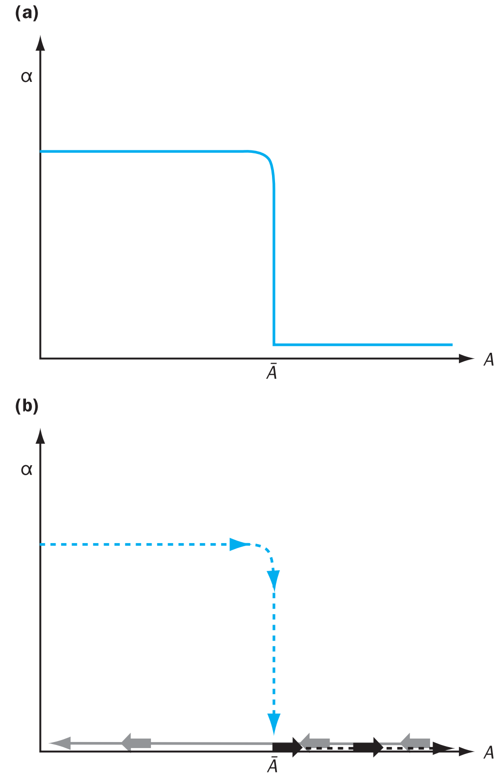

Figure 5.6 Threshold effects and irreversibilities

Another way of thinking about this issue is in terms of carrying capacities (or assimilative capacities, as they are sometimes called) of environmental media. In the case of water pollution, for example, we can think of a water system as having some capacity to transform pollutants into harmless forms. The stock pollution model examined in this section has in effect assumed unlimited carrying capacities: no matter how large the load on the system, more can always be carried and transformed at a constant proportionate rate. Whether this assumption is plausible is, in the last resort, an empirical question. Where it is not, modelling procedures need to reflect limits to carrying capacity. The suggestions for further reading point you to some literature that explores models with variable pollution decay rates.

5.11 Departures from convexity or concavity in damage and abatement cost (or pollution benefit) functions

When pollution benefit and damage functions were first presented in Section 5.4, a number of assumptions were made about their shapes which guaranteed that each was 'well behaved'. In particular, it was assumed that the total damage function was convex and that the total benefit function was concave. These give rise to the shapes of the total functions shown in the upper part of Figure 5.2. This in turn implies that the marginal damage function is upward sloping and the marginal benefit function is downward sloping.[18] After explaining the notions of convexity and concavity, we focus attention on the damage function, giving some examples of why damages may not be convex, and show some consequences of non-convexity in pollution damage.

Consider a function, $f(x)$, of a single variable x. The function is *strictly convex* if the line segment connecting any two distinct points on the function lies everywhere above the function $f(x)$, except at the two points themselves. The function is convex (as opposed to strictly convex) if the line segment lies everywhere above or on the function $f(x)$, but not below it.[19] As an example, the function graphed in Figure 5.7 is strictly convex. Looking back at Figure 5.2, it is clear that the damage function $D(M)$ is convex.[20]

With a simple change of words, we obtain equivalent definitions for concavity. Thus, a function, $f(x)$, of a single variable x, is *strictly concave* if the line segment connecting any two distinct points on the function lies everywhere **below** the function $f(x)$, except at the two points themselves. A function is concave (as opposed to strictly concave) if the line segment lies everywhere below or on the function $f(x)$, but not above it. Therefore, the B(M) function in Figure 5.2 is strictly concave. Thus far we have been referring to functions. But the notions of convexity and non-convexity also apply to sets. A set is convex if and only if, for any two elements belonging to that set, every point on the line joining those two elements also lies in the set. Moreover, the set is strictly convex if such a line everywhere lies inside the set rather than on its boundary. Figure 5.8 illustrates. Note that if a function is convex the set of points that lies above the function is a convex set, as shown in Figure 5.9.

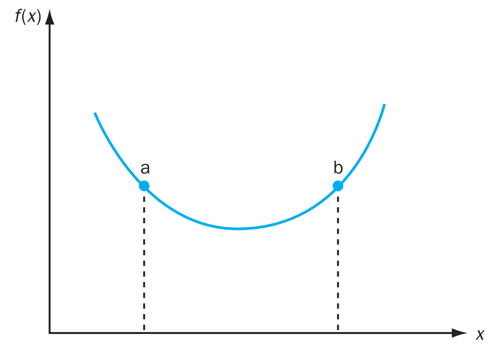

Figure 5.7 A strictly convex function

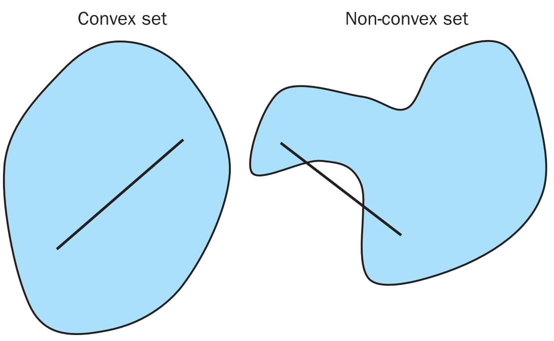

Figure 5.8 Convex and non-convex sets

[18] It should be noted that linearity of the marginal functions (as drawn in the lower part of Figure 5.2) is *not* implied by convexity or concavity alone. Those marginal functions were portrayed as linear for convenience only. However, convexity does imply that the marginal damage function is nowhere downward sloping nor discontinuous, and the marginal benefit function is nowhere upward sloping nor discontinuous.

[19] For simplicity, our definitions are given for functions of a single variable *x*. But they are readily generalisable to functions of more than one variable. See, for example, Layard and Walters (1978).

[20] In fact, as drawn it is strictly convex. But what matters is whether the weaker property of convexity is satisfied. So we shall use the word 'convex' from now on to cover strict as well as (weak) convexity.

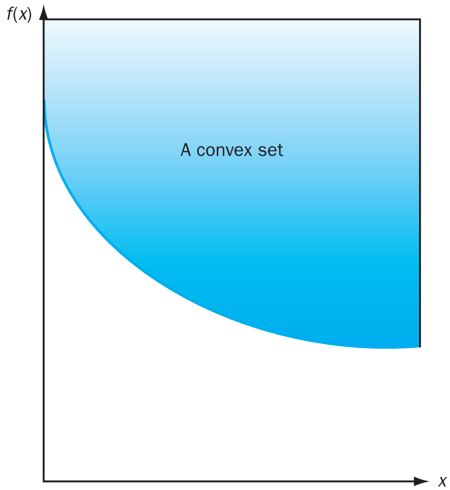

Figure 5.9 A convex function, $f(x)$, generating a convex set defined by points lying above it

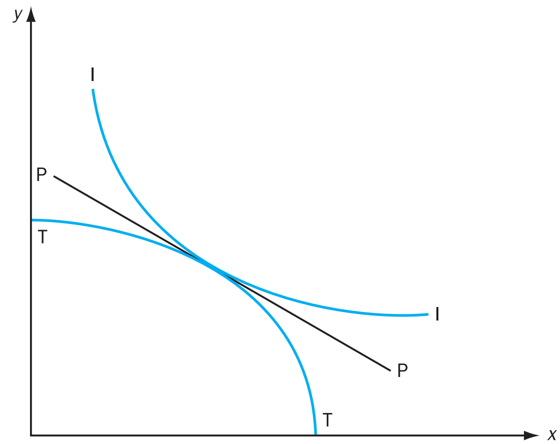

Figure 5.10 (Strict) convexity generates equilibrium with a unique set of relative prices

Why is convexity important in economics? The answer is that where standard convexity conditions are satisfied we obtain equilibria that satisfy optimising objectives. When strict convexity is satisfied, the equilibrium is unique. The most important application of this idea in economics is to the second theorem of welfare economics, which asserts that:

> ***If*** *production possibilities are convex and continuous* ***and*** *each individual has a continuous utility function that generates indifference curves such that the area above any indifference curve is a convex set* ***then*** *each possible Pareto optimum can be achieved as a competitive equilibrium, with an appropriate distribution of initial endowments.*

Figure 5.10, adapted from Mäler, 1985, illustrates this intuitively. An individual can produce two goods x and y. Her production possibilities are represented by the convex set given by the transformation curve TT and the area below it. The individual has a utility function that generates convex indifference curves, one of which is shown in Figure 5.10, labelled as II. The tangency between TT and II generates a straight line that gives the relative price of y in terms of x.

The discussion above shows that what is really important in all this is that the functions describing the problem being investigated are smooth, continuous, and lead to unique marginal efficiency conditions. These properties are satisfied by the well-behaved benefit and damage functions used in Figure 5.2. It is clear from the lower panel of Figure 5.2 that there is just one level of pollution at which the marginal efficiency first-order condition is satisfied: the marginal benefit of pollution (or equivalently marginal cost of abatement) is equal to the marginal damage of pollution. Moreover, with convex damage and concave benefits, second-order conditions will necessarily be satisfied too. This implies that marginal analysis alone is sufficient for identifying the efficient level of pollution.

5.11.1 Non-convexity of the damage function and its implications

There are many reasons why the damage function may be non-convex or the pollution benefit function may be non-concave. Here we restrict attention to the more commonly discussed case of non-convex damages. So what might cause a pollution damage function to not be of the smooth, continuously increasing form that we have assumed so far? One example was given implicitly in Section 5.10 where we introduced the ideas of threshold effects and irreversibility. A closely related example is acidic or eutrophic pollution of rivers and lakes. Here, pollution may reach a threshold point at which the lake becomes biologically dead, unable to support animal or plant life. No further damage is done as pollution levels rise beyond that point. The total and marginal

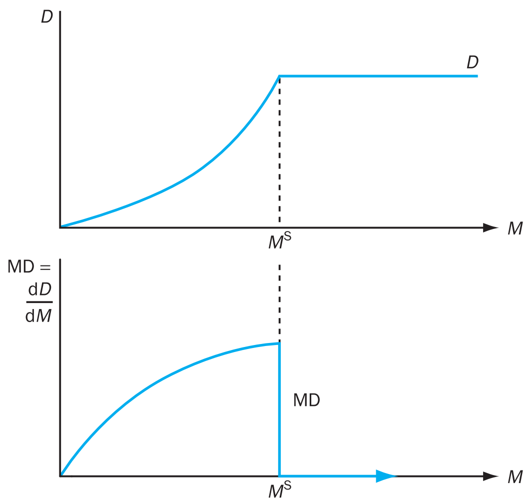

Figure 5.11 A non-concave damage function arising from pollution reaching a saturation point

damages functions in this case will be of the form shown in Figure 5.11.

The lake pollution example is one instance of what some authors mean when they say that 'nature is not convex'. Another case, discussed in Fisher (1981), is non-convexity of damages arising from averting behaviour by individuals, in which case the non-convexity arises from economic behaviour. Suppose a factory emits particulate emissions that create external damages for an individual living in the neighbourhood of the factory. The marginal damage initially rises with the amount of the pollution. However, at some critical level of pollution flow, the affected individual can no longer tolerate living in the neighbourhood of the factory, and moves to live somewhere else where the damage to him or her becomes zero. As far as this particular individual is concerned, his or her marginal damage function is also of the form shown in Figure 5.11. However, if there are many individuals living in the neighbourhood, with varying tolerance levels, many of whom are prepared to move at some level of pollution, the aggregate marginal pollution damage function will be the sum of a set of individual functions, each of which has the same general form but with differing pollution tolerance levels. The aggregate damage function will be of the non-convex form shown in the top panel of Figure 5.12, with its marginal counterpart being shown by the curve labelled MD in the central panel.

Now combine the marginal damage function for the averting-behaviour example with a marginal benefit function of conventional shape. This is shown in the central panel of Figure 5.12. Marginal damage and benefits are equalised here at three emission levels. Marginal analysis alone is no longer sufficient to identify a single efficient target.

Box 5.5 More on the lake pollution example

The non-convex lake pollution damages illustrated in Figure 5.11 is in essence identical to the shallow-lake eutrophication example examined by Carpenter *et al.* (1999). There, the authors provide a theoretical foundation for the convexity in terms of the ecology of shallow lakes subject to phosphorus (P) loading. What is striking about their analysis is that under particular conditions (including prolonged exposure to P-inputs and high recycling of P-sediments into the lake waters) the damages done cannot be reversed merely by reducing the flows of P-bearing nutrients into the lake ecosystem. Therefore, conventional tax- or permit-based environmental policy instruments – which would operate by reducing external loadings of pollutants into the lake – may be unable to achieve the task of restoring the lake to an economically efficient level of water quality. Instead, direct 'clean-up' interventions (such as oxygenation, mebiomanipulation, and aluminium sulphate treatment) are required to shock the system to a new stable equilibrium in which the lake is no longer biologically dead. However, such interventions may not always be able to reverse the damage. Should direct intervention be successful, then after that has been achieved conventional tax- or permit-based environmental policy instruments may be sufficient to maintain lake water quality. This case is also discussed more briefly (and both simply and elegantly) in Arrow *et al.* (1999).

The shallow lake eutrophication example also shows that non-convexity can have important implications for the choice of instrument that policy makers adopt to attain pollution targets, and so we shall return to this example in the following chapter.

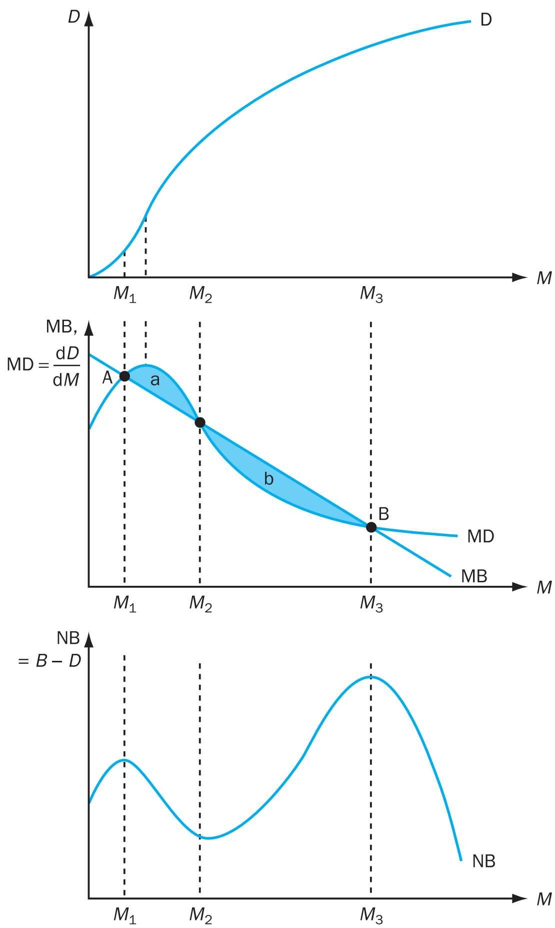

Figure 5.12 Multiple marginal equalities arising from a non-convex damage function: the case of behavioural adjustments of individuals

To ascertain which of these, if any, is the efficient level of pollution, it is necessary to inspect the level of total net benefits at these three points, *and* at all other levels of emission (as maximised net benefits will not necessarily even correspond to a marginal equality when one or more function is not convex). The two points labelled A and B are 'local optima', as they satisfy the second-order condition for a local maximum of net benefits, as shown in the lower panel of Figure 5.12. In this case it can be seen by inspection of the NB curve that M_3 is a 'global' net benefits-maximising pollution level. Note that in moving from M_1 to M_3, net benefits at first fall (by the area labelled **a**) and then rise (by the area labelled **b**).

Convexity matters, therefore, because in its absence conventional first-order conditions for a maximum (nor even second-order conditions) will not necessarily help us to identify welfare-maximising pollution targets. But this is not just a mathematical quirk; it is also of considerable import for public policy. There are three major reasons why. The first could be described as a 'practical' matter: calculating the efficient level of emissions (or pollution stock) will in general be more complicated than where all functions satisfy convexity or concavity as appropriate. This is partly about computational difficulty. But more importantly, it is to do with the fact that the information required to identify the non-well-behaved functions may be very costly to obtain. Obtaining reliable estimates of functions will be particularly difficult where information is limited or uncertain (a matter discussed further in Chapter 7).

Secondly, non-convexity may be important because it exists but we do not recognise that it exists. In that case, some commonly advocated tools could give seriously misleading results. For example, a failure to recognise the existence of threshold effects or irreversibilities could render project appraisal using cost–benefit analysis completely wrong. (An example is explored in Problem 5 at the end of this chapter.) One reason why policy makers may fail to recognise non-convexity is to do with the way information is acquired. We often find out about things by exploring a relevant 'local neighbourhood'. For example, cross-section sampling techniques may generate data on emissions and damage that are clustered around current levels, and tell us little or nothing about properties of the function outside the current sample range. Inspection of that data may suggest convexity when in fact the function is only convex over part of its range. This becomes important – and potentially dangerous – if the policy maker falsely projects the apparently convex function outside this current range.

A third reason – which we develop at some length in the next chapter – is that non-convexity not only affects the choice of pollution target but has implications too for the choice of policy instrument through which we seek to achieve our target.

Our final example of the potential importance of non-convexity concerns the likelihood that economically superior energy technologies may not be adopted because of lock-in of existing technologies. This is discussed in Box 5.6.

Box 5.6 Non-convexity, lock-in and the choice of energy technology

In a series of articles, Robin Cowan and David Kline argue that positive feedback (or 'lock-in'), which has its roots in various forms of increasing returns to scale, can lead to economically inferior choices of environmental technology. Rational market behaviour can lead to sub-optimal outcomes in which more cost-effective technologies remain unselected. Moreover, conventional environmental policy using quasi-market incentive-based mechanisms can reinforce this lock-in of prevailing techniques.

Underlying these problems is non-convexity in net benefit (or profit) functions. As with all cases of non-convexity in environmental economics, this has implications both for the targets that are chosen and the instruments that are selected to attain those targets. With respects to targets, Cowan and Kline argue that lock-in means that we may aim for the wrong thing. An important example is the lock-in of prevailing fossil fuel energy technologies (and the corresponding lock-out of emerging renewable energy technologies). Here, the target may be wrong in a qualitative sense (the wrong technology is being selected) rather than in a quantitative sense (we are aiming for too much or too little of something). With respect to instruments, Kline (2001) shows how SO_2 trading systems can encourage end-of-pipe emission reduction solutions when a variety of other pollution prevention strategies would be both economically and environmentally superior.

Let us consider the analysis of technology lock-in a little more deeply. Figure 5.13 illustrates. The horizontal axis is labelled 'Location in Economic Space' and refers to some relevant dimension for the particular problem being studied; it might, for example, refer to the relative labour intensity of a production process. In our case, suppose that it refers to energy technologies ranked in terms of carbon emissions per unit of energy produced. The vertical axis refers to the total cost of producing a given amount of energy from the available technologies.

The left picture portrays a conventional 'neoclassical' world in which choices satisfy convexity conditions. Suppose that the position of the dark ball denotes the location in space of the currently adopted technology. Three processes would all lead to the adopted technology converging over time towards a least-cost optimal solution. First, a 'local' search process in which firms estimate the relative costs of making marginal changes to technology that alter carbon emissions. Second, government policies (such as emissions taxes or tradable permits) that generate incentives on firms to make marginal changes in techniques or emission control technologies. Third, a series of shocks could disturb the location of the dark ball; firms then react by accepting the change or moving back to the original location.

But now suppose that the economic environment is one as portrayed by the right-hand side picture. It is clear that there are now two local minima. However, the global cost minimum is not one to which the system will be attracted under any of the three kinds of processes referred to above. Instead, given the starting point shown in the picture, the system will tend to be attracted towards and then 'locked into' the less efficient outcome.

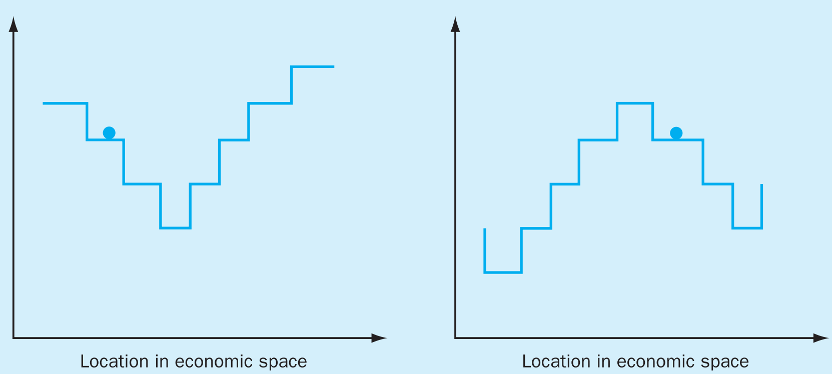

Figure 5.13 Visualising lock-in

Box 5.6 *continued*

Notice the importance of initial conditions to this argument. Where the economy finishes up here depends critically on where it starts off; there is path dependency. It may well be that one technology happens to be the cheapest at some point in time, and firms rationally pursue this approach. As time goes by, learning benefits, economies of scale, and network externalities may well reinforce the cost advantage of this first-chosen technology, leading to lock-in. However, the globally superior technology may be quite different – but to teach it is very difficult, and might even seem value-destroying when looking at matters from a local perspective.

The authors argue that the methods of getting from one inferior technology to another superior technology often require policy instruments quite different from the conventional market-based instruments that economists typically advocate. We return to this matter in the next chapter.

References: Cowan (1990, 1991); Cowan and Kline (1996); Kline (2001)

5.12 'No regrets' policies and rebound effects

It is sometimes argued that environmental objectives can be achieved at no cost or, better still, at 'negative' overall cost. Such options are known as 'no regrets' policies. There are several reasons why these may arise. First, the economy may be one in which there are currently technical and/or economic inefficiencies in the energy-using or energy-producing sectors. The existence of market imperfections can cause firms to be producing away from the frontier of what is technically and/or economically possible. Firms may be unaware of new techniques, or poorly informed about waste recycling mechanisms. Companies may have old, technologically obsolete capital, but are unable because of credit market imperfections to update even when that would generate positive net present value. Given such inefficiencies, an environmental programme that requires firms to use new, less polluting techniques, or which provides incentives to do so, can generate a double benefit. Pollution is reduced and productive efficiency gains are made. Moreover, under some circumstances those joint benefits may more than offset the present value of the resource costs of the environmental programme.

The validity of such arguments depends not only on the existence of such inefficiencies but also on the property that the transactions and coordinating costs of removing them would not exceed the efficiency gains achieved. Furthermore, if subsidies are used as incentivising devices, there are implications for public finance, and one would also need to take into consideration any new distortions arising from the financing of those subsidies.

There may be other problems with energy efficiency enhancement projects. If done on a widespread basis, the reduced demand for energy will decrease the price of energy, thereby generating a substitution effect towards more energy-intensive techniques. This would offset some of the initial gains and could, in some circumstances, lead to increased energy use (and so, possibly, an increase in associated total emissions). This 'rebound effect' is considered in Grant *et al.* (2007) and discussed in Chapter 8.

More generally, any environmental improvement programme that has joint benefits could be one that has a negative net cost (or positive net benefit) if the value of the joint benefits exceeded the gross costs of the programme. For example, consider a road pricing scheme the principal goal of which is to reduce traffic congestion and so reduce total travel time. Lower traffic congestion would likely also reduce vehicular emissions (such as particulates, sulphur dioxide, nitrogen dioxide and carbon monoxide), thereby generating additional health benefits. One would also expect there to be visual amenity, noise reduction, and road traffic accident reduction benefits. Hence in appraising any particular environmental improvement programme, it is important to make sure that all significant costs and benefits are considered.

Some of those benefits may accrue from dynamic efficiency gains, and so accrue over long periods. One special case of this is dynamic efficiency gains, arising through induced technical change. It has long been recognised (see, for example, Porter, 1991) that some forms of regulatory constraint may induce firms to be more innovative. If a pollution control mechanism can be devised that accelerates the rate of technical change, then the mechanism may more than pay for itself over the long run. One area where this may be very important is in policy towards the enhanced greenhouse effect. Grubb (2000) argues persuasively that the provisions of the Kyoto Protocol will have beneficial induced effects on technical change. He writes:

> general economic processes of international investment and the dissemination of technologies and ideas – accelerated by the provisions on technology transfer and other processes under the Convention and the Protocol – could contribute to global dissemination of cleaner technologies and practices. In doing so, they will also yield multiplicative returns upon industrialised country actions.
>
> *Grubb (2000), p. 124*

5.13 The double dividend hypothesis

Some of the reasoning in the previous section implied that emission reductions programmes may create joint benefits, and that when jointness is taken into account the net welfare impacts may be substantially enhanced. Beginning in the 1990s, an important literature has developed that has formalised this idea by considering environmental improvement programmes within the context of a package that also includes reform of the structure of the taxation system. This literature has come to be known by one of its key propositions, the double dividend hypothesis. This hypothesis arises from the possibility that the revenues from an emissions tax (or a system of permits sold by auction) could be earmarked to reduce marginal rates of other taxes in the economy. If those other taxes have distortionary (i.e. inefficiency-generating) effects, then reducing their rate will create efficiency gains. Thus an environmental tax with revenues ring-fenced for reducing distortionary taxes has a double benefit (or double dividend); the environment is improved *and* efficiency gains accrue to the economy as whole.

In order to get a good understanding of the double dividend hypothesis, and the circumstances in which the hypothesis may be true, one needs to be clear about the framework in which this issue is normally analysed. The analysis in previous sections of this chapter has implicitly assumed a benchmark or reference case of a 'first-best world' in which there would be a complete competitive equilibrium other than for the presence of one single market distortion. That distortion consists of an external effect: there exist pollution emissions that are not compensated for through a market mechanism. As we shall see in the next chapter, if an emissions tax were levied on pollution generators at an appropriate rate (or there was a system of marketable emission permits with an appropriate quantity of permits being issued) then the internalisation of this externality would shift the economy to the fully efficient competitive equilibrium. This kind of analysis – with which much of this and most other economic textbooks deal – is 'first-best' analysis.

Let us consider Figure 5.3 once again. The diagram corresponds to, and is strictly valid only in, such a first-best setting. It takes one market alone – the one in which there is a market failure – and deduces what would be the economically efficient level of output of the product in this market (or in this particular context, the efficient level of the emissions associated with that output level). Most importantly, it does so conditional on the assumption that in the rest of the economy all other markets satisfy resource allocations that correspond to full competitive equilibrium.

Thus, taking the lower part of Figure 5.3, in the absence of internalisation of the externality, the emissions level would be $\hat{M}$. However, if a Pigovian tax at rate μ^* per unit emission were levied on the externality generator, then emissions would fall to their first-best level, M^*. The efficiency gain that results from the introduction of the Pigovian tax is given by area B. (Moving from $\hat{M}$ to M^* reduces the value of output net of direct production costs by the amount A, the integral of the area under the marginal benefit curve between $\hat{M}$ and M^*. However, it also reduces the value of environmental damage by the amount A + B, the integral of the area under the marginal damage curve between $\hat{M}$ and M^*. So the net efficiency gain is (A + B) – A = B. However, as we shall see in a moment, this conclusion is not generally valid outside of a first-best world, as the area A will typically misrepresent the welfare value of the net output lost.

Once we are outside a first-best world we are in a second-best one in which there are multiple distortions from first-best competitive equilibrium. In reality, this is the kind of world we actually live in: one in which there are many market failures or other distortions occurring together (involuntary unemployment, public goods produced at less than optimal levels, various kinds of distortionary taxes, and so on).

In a second-best world the impacts of any kind of policy intervention will be context specific, and so it is difficult or impossible to obtain general results about what will be the welfare-maximising level of any single policy target or intervention. To derive policy-relevant results, one has to specify a particular set of benchmark 'second-best' circumstances and then analyse the consequences of particular interventions relative to that benchmark.

What much of the double dividend research literature has done can be roughly characterised as the following:

1. Begin by identifying one environmental problem in which we are particularly interested, such as an uncompensated pollution externality. It is this externality on which an emission tax will be levied.
2. Specify the appropriate second-best benchmark economy, making clear exactly which particular configuration of inefficiencies or distortions are assumed to prevail in that economy. For the double dividend hypothesis to be meaningful there must be at least two distortions in the economy, one being the environmental problem referred to in 1. above, and a second being a distortionary tax. There can of course be more than two distortions in total, such as a variety of distortionary taxes, imperfect labour markets, or involuntary unemployment.
3. One distortionary tax is then focused upon. The revenues from the polluting emissions tax will be recycled so as to reduce the marginal rate of this distortionary tax. In early studies, this tax was typically chosen to be a tax on labour income, but it could be any (non-lump-sum) tax.
4. With the model of the economy having now been specified, one solves that model for its optimal welfare-maximising solution twice: once in the benchmark case in which there is no policy intervention; second in the simulation run case in which an emissions charge is imposed on the pollution externality, and the revenue obtained from that charge is used in its entirety to reduce the selected distortionary tax.
5. Calculate the difference in welfare between the case of no intervention – the *status quo ex ante* benchmark – and the case in which the emissions charge is introduced with the revenue recycled to reduce the distortionary tax.

The schema that has just been outlined is representative of the way that research is typically done in this field. There are many variants of it. For example, changes in welfare may not be the only criterion used for comparing baseline and intervention cases. One might also be interested in impacts on unemployment, real wages, or GDP, for example. Second, one might choose to compare several alternative uses of the revenue from the environmental tax rather than just the one alternative mentioned above. This way, one can gain some insight into the relative advantages of alternative ways of using those revenues. And – as is in fact usually the case – one might choose to ignore the environmental improvements themselves (i.e. the first dividend) and instead just measure the net welfare (or other) changes that come from all other impacts of the intervention (the second dividend, if you like).

To get a flavour of the kind of results that have emerged from the literature, we start with a key result established by Bovenberg and de Mooij (1994b, 1994a), in which they analyse a tax reform in which an environmental tax is introduced the revenues from which are used to finance reductions in a tax on labour income. Under what circumstances would this policy yield an overall increase in non-environment-related welfare (i.e. an increase in economic welfare without including the gross benefits of the environmental improvement that results from the environmental tax)? Bovenberg and de Mooij conclude that there will be an overall increase in non-environment-related welfare if and only if the uncompensated (Marshallian) wage elasticity of labour supply is negative (i.e. the labour supply curve is backward bending). But as most empirical studies find this elasticity to be positive, this 'strong form' of double dividend hypothesis is rejected.

Goulder (1995) provides some intuition for this result. He writes:

> There are two components. First, the tax on the environmentally harmful consumption good lowers the after-tax wage and generates distortions in the **labor** market, and these labor-market distortions are at least as great in magnitude as the labor market distortions from a labor-tax increment of equal revenue yield. Hence the revenue-neutral swap in which the environmental tax replaces (some of) the labor tax leads to no reduction (and usually an increase) in labor-market distortions. Second, the tax on the environmentally damaging commodity induces changes in the commodity market – 'distorting' the choice among alternative commodities.

What is going on here is that the policy has effects working in opposite directions:

1. Welfare is **increased** as the distortionary tax on labour income is reduced. This is known as the 'revenue-recycling' effect.
2. Welfare is **decreased** as the environmental tax increases the price of that good, reducing the after-tax wage, and creating distortions in the labour and commodity markets. This is known as the 'tax-interaction effect'.

Moreover, under the plausible condition pointed out by Bovenberg and de Mooij (that the labour supply wage elasticity is positive), the tax-interaction effect dominates the revenue-recycling effect, and so the strong form of double dividend hypothesis is rejected. Note, however, that even if the overall (non-environmental) welfare impacts were negative, then it may still be the case that overall economic welfare would increase. For that to happen, the benefits of the environmental improvement would need to be a high enough size.

The notions of revenue-recycling and tax-interaction effects can help us to explain further a point we made earlier regarding the partial equilibrium analysis of an emission tax. In discussing Figure 5.3 it was pointed out that in general the area that we labelled above as A misrepresents the welfare cost of the environmental tax. It does so because:

1. to the extent that there is a revenue-recycling effect, area A overstates the welfare cost;
2. to the extent that there is a tax-interaction effect, area A understates the welfare cost.

Therefore, the Goulder and Bovenberg/de Mooij results imply that the true second-best welfare costs are higher than indicated in Figure 5.3. This, in turn, implies that the second-best environmental tax should be less than the (first-best) Pigovian tax indicated there.

How general are the Goulder and Bovenberg/de Mooij results?

We remarked earlier that it is difficult or impossible to obtain general results using second-best analysis. This suggests that one should be very wary of taking the conclusions above as being valid in all conceivable circumstances. Such caution is certainly warranted. The Goulder and Bovenberg/de Mooij results are obtained under a particular set of simplifying assumptions about the nature of the economy. These assumptions deal with such things as how many and what kinds of inputs enter into production; how many and what kind of subsidies and taxes currently exist; and the numerical values of the model's parameter values. Changing any of these will change results, and may do so very substantially (for example, changing the sign of the tax-interaction effect and so certainly leading to double dividends).

Also of importance is treatment of unemployment. The Goulder results are derived from a model in which all markets clear. But introducing the prior presence of involuntary unemployment can (but does not necessarily) alter the qualitative conclusions regarding welfare changes. Once a world in which involuntary unemployment is specified as part of initial conditions, the question is raised of whether welfare changes are the most appropriate criterion for assessing policy interventions (or whether one might also wish to use a variety of other criteria). Plausible alternative (or additional) candidates here include employment or output impacts.

One interesting example here is Bonetti and FitzRoy (1999). These authors formulate a model in which there is an imported polluting resource and prior involuntary unemployment. They consider the effects of taxes on energy, holding the real wage constant, under differing levels of government expenditure and externalities. Potential net welfare and employment gains are found to be substantial for plausible parameters, although their simulations reveal conflict between the goals of net welfare, employment and profitability over much of the relevant parameter range. They also find that the optimal energy tax is less than the Pigovian tax for plausible externalities.

What conclusions can we derive relating to the double dividend debate? In the very early days, analyses of revenue-neutral tax reforms in which environmental tax revenues were used to reduce taxes on labour income often reached very optimistic conclusions in which the reform would generate welfare gains even in the absence of the environmental gains. All that an advocate of such a policy would need to establish is that the environmental benefits were at least positive.

But such results sometimes ignored tax-interaction effects – the additional distortions induced by the environmental tax itself in a second-best world. The Bovenberg and Goulder results brought tax-interaction impacts into consideration, and established that in simple market-clearing models, double dividends were unlikely to be available under plausible parameter assumptions. Once again, though, it is important to be aware that the Goulder and Bovenberg/de Mooij results are obtained under a particular set of simplifying assumptions. Thus, they depend not only on parameters but also on the form and arguments of the utility functions (for example, labour is all disutility, there are no interdependencies) and that it is just *current* welfare that is considered.

Recent research has pointed to sets of additional sets of circumstances in which double dividends would be realised, particularly when relevant criteria are extended to include employment impacts. (See, for example, Table 8.14 in Chapter 8 of this book.) Finally, we should not lose sight of the fact that if an environmental tax brings about sufficiently large environmental benefits, then even where there are increased distortions elsewhere in the economy the tax package could raise overall economic welfare, whilst possibly reducing both pollution and unemployment.

Before leaving this discussion about possible double dividends, it is worth noting that at any moment in time there are invariably opportunities to reform the tax system to make it less distorting and so more efficient. Most of these reforms do not involve environmental policy. It can also be shown that there are circumstances in which one would want to distort environmental policy by increasing or decreasing the environmental tax rate (as appropriate) in order to raise revenue when the marginal cost of raising revenue elsewhere is greater. Hence, the double dividend argument is best regarded as one special case of a more general class of packages that can increase the efficiency of public finance and expenditure.

5.14 Objectives of pollution policy

We conclude this chapter by returning to an issue addressed at its outset: the objectives of pollution policy. Throughout most of this chapter a single criterion, economic efficiency, has been adopted. But it would be both naïve and wrong to assume that this is, or should be, the only or even the main objective. Each of the following has some claim to be regarded as a goal of environmental policy, and so might be given some weight in determining pollution targets with regard to stock pollutants:

- sustainability and ecological goals (e.g. minimum disruption to ecosystems, maintenance of biodiversity);
- human health protection;
- public preferences.

We now give some brief consideration to each of these.

Sustainability and ecological goals

Many people feel passionately that sustainability should be the principal environmental policy objective. It is also a stated goal of many national and international official organisations, of some international environmental agreements, and of many NGOs. There are other people who would argue that specific ecological objectives – such as avoiding decline in biodiversity, or minimising threats to ecosystem resilience – should be that principal goal.[21] Economics cannot tell us anything useful about what *should be* an ultimate policy goal. Questions about what should be the case are ethical questions, and at the societal (as opposed to personal) level can only be resolved through social choice or political processes.

However, economics may be able to tell us something useful about what would be the consequences of pursuing particular goals or objectives. As far as sustainability is concerned, one cannot get very far without establishing a precise meaning for that term. In the sense that we use the term in this book (non-declining utility per capita over indefinite time) it is not evident that sustainability itself has anything to say about pollution targets. We have developed a modelling framework in this chapter in which pollution targets can be chosen to maximise some suitably defined measure of net economic benefit. But as we show elsewhere in (particularly in Chapters 16 and 19) that the time profile of consumption that emerges from this optimisation exercise will not typically be a sustainable path (in the sense of constant utility, or constant consumption, over indefinite time). Assuming that sustainability is feasible, a sustainability constraint might then be imposed which would alter the consumption time profile to the one giving maximum possible constant consumption. This would have a lower present value of net benefits than the unconstrained optimal consumption path; but that, of course, would simply be the price of imposing the sustainability constraint. But the key point here is that none of this would alter the pollution target choices previously made; the ones that maximise the present value of unconstrained net benefits would remain operative.

With respect to what may well be the worst case, the Stern Review says, in effect, that '*business as usual*' climate change implies that utility is lower than it would otherwise have been but not lower than it is now; even in its worst-case scenario the Review has utility growth continuing. Interestingly, Stern in effect adopts modified a safe minimum standard: the Review contends that policy makers should adopt a maximum atmospheric CO_2 concentration threshold of 550 ppm, and that not crossing that threshold would not be very expensive.

However, we have not yet paid any serious attention to irreversibility, non-sustainability situations, thresholds and uncertainty. We shall do so in Chapter 7, and at various places later in the book. Once these matters are brought into consideration, then, as you will see, sustainability considerations might play a more central role in pollution target choices.

Health

Pollution targets are in practice often determined (or at least heavily influenced by) perceived risks to human health. This is evident in the setting of standards for background radiation levels around nuclear energy facilities, and for standards for localised pollutants (such as ozone levels or particulate matter) in urban areas. Some other examples are given in Tables 5.3 and 5.4. In the USA, the 1990 Clean Air Act requires the US EPA to set *National Ambient Air Quality Standards* for pollutants considered harmful to public health and the environment. The legislation distinguishes between two types of national air quality standards: *primary standards* set limits to protect public health, including the health of 'sensitive' populations such as asthmatics, children, and the elderly; and *secondary standards* set limits to protect public welfare more generally, including protection against decreased visibility, damage to animals, crops, vegetation, and buildings. So while these controls are principally health-oriented, they indicate the multi-functionality of many pollution targets.

There may, of course, be no safe level of pollution given the heterogeneous nature of the health of the

[21] One illustration of this is the United States *Endangered Species Act*, which imposes ecological sustainability standards and mandates protection of designated species **at any cost**. Compare this with two other US efficiency-based statutes, the *Federal Insecticide, Fungicide and Rodenticide Act* (FIFRA) and the *Toxic Substances Control Act* (TSCA), which restrict the use of agricultural chemicals and other dangerous or toxic substances, and for which targets are to be set on efficiency grounds, with the requirement that the benefits and costs of regulation are to be balanced in both cases.

Table 5.3 Environmental targets that are largely determined by human health considerations

Pollutant	Target	Relevant criterion
United Kingdom		
Cadmium/lead	Discharges into North Sea to fall by 70% between 1985 and 1995	Health criterion
Sewage concentration	Max. 30 mg/litre suspended solids: Max. Biological Oxygen Demand (BOD) 20 mg/litre	1976 National Water Council: perceived health risks (the regulation is also justified in terms of the precautionary principle)
Asbestos	The Control of Asbestos Regulations 2006. Prohibition of asbestos, control of asbestos at work and asbestos licensing. Prohibition of importation, supply and use of all forms of asbestos. Ban the second-hand use of asbestos products such as asbestos cement sheets and asbestos boards and tiles; including panels which have been covered with paint or textured plaster containing asbestos. If existing asbestos-containing materials are in good condition, they may be left in place, their condition monitored and managed to ensure they are not disturbed. Mandatory training for anyone liable to be exposed to asbestos fibres at work. When work with asbestos or which may disturb asbestos is being carried out, Regulations require employers and the self-employed to prevent exposure to asbestos fibres. Where this is not reasonably practicable, they must make sure that exposure is kept as low as reasonably practicable by measures other than the use of respiratory protective equipment. The spread of asbestos must be prevented. Worker exposure must be below the airborne exposure limit, for all types of asbestos, of 0.1 fibres per cm^3. A Control Limit is a maximum concentration of asbestos fibres in the air (averaged over any continuous 4-hour period) that must not be exceeded. Short-term worker exposure should not exceed 0.6 fibres per cm^3 of air averaged over any continuous 10-minute period using respiratory protective equipment if exposure cannot be reduced sufficiently using other means. Respiratory protective equipment must be suitable, must fit properly and must ensure that worker exposure is reduced as low as is reasonably practicable.	
PCBs	Phase out by 1999	Strict precautionary principle – health risks
United States		
Criteria air pollutants	Clean Air Act (CAA) (as amended 1990) www.epa.gov/oar/oaqps/peg_caa/pegcaain.html National ambient air quality standards for pollutants considered harmful to public health and the environment. For details, see Table 5.4.	Health risks (but subsequently broadened to also include environmental damage risks). Standards to be set on safety grounds (to achieve an 'adequate margin of safety'). US EPA must consider benefits of regulation but *not* costs
Water Quality Clean Water Act (CWA) 1987 (in conjunction with Federal Water Pollution Control Act) www.epa.gov/region5/defs/html/cwa.htm	Effluent emissions, from stationary-point sources and non-point sources	Standards to be set on safety grounds. Waters required to be at least 'swimmable and fishable'. US EPA must consider benefits of regulation but *not* costs
Hazardous waste disposal on land, both current disposal (RCRA) and abandoned waste dumps (Superfund)	The Resource Conservation and Recovery Act (RCRA). Comprehensive Environmental Response, Compensation, and Liability Act (Superfund). The Superfund Amendments and Reauthorization Act (SARA) www.epa.gov/superfund	Standards to be set on safety grounds US EPA must consider benefits of regulation but not *costs*
International		
CFCs	CFC production to fall to 80% and 50% of 1986 levels by 1994 and 1999 respectively	Concept of 'critical load', chosen to limit ozone depletion to levels not significantly threatening human health (although health considerations are not the unique criterion)

Key: mg/litre: Milligrams per litre of water. 'Critical load' and 'precautionary principle' are explained elsewhere in the chapter.
Source: Asbestos: UK HSE. http://www.hse.gov.uk/asbestos/regulations.htm (accessed 18 September 2008)

Table 5.4 National Ambient Air Quality Standards (NAAQS) for Criteria Air Pollutants, USA, 2008

	Primary Standards		Secondary Standards	
Pollutant	Level	Averaging Time	Level	Averaging Time
Carbon Monoxide	9 ppm (10 mg/m^3)	8-hour[1]	None	
	35 ppm (40 mg/m^3)	1-hour[1]		
Lead	1.5 μg/m^3	Quarterly Average	Same as Primary	
Nitrogen Dioxide	0.053 ppm (100 μg/m^3)	Annual (Arithmetic Mean)	Same as Primary	
Particulate Matter (PM_{10})	150 μg/m^3	24-hour[2]	Same as Primary	
Particulate Matter ($PM_{2.5}$)	15.0 μg/m^3	Annual[3] (Arithmetic Mean)	Same as Primary	
	35 μg/m^3	24-hour[4]	Same as Primary	
Ozone	0.075 ppm (2008 std)	8-hour[5]	Same as Primary	
	0.08 ppm (1997 std)	8-hour[6]	Same as Primary	
	0.12 ppm	1-hour[7] (Applies only in limited areas)	Same as Primary	
Sulfur Dioxide	0.03 ppm	Annual (Arithmetic Mean)	0.5 ppm (1300 μg/m^3)	3-hour[1]
	0.14 ppm	24-hour[1]		

Units of measure for the standards are parts per million (ppm) by volume, milligrams per cubic meter of air (mg/m^3), and micrograms per cubic meter of air (μg/m^3).
[1] Not to be exceeded more than once per year.
[2] Not to be exceeded more than once per year on average over 3 years.
[3] To attain this standard, the 3-year average of the weighted annual mean PM2.5 concentrations from single or multiple community-oriented monitors must not exceed 15.0 μg/m^3.
[4] To attain this standard, the 3-year average of the 98th percentile of 24-hour concentrations at each population-oriented monitor within an area must not exceed 35 μg/m^3 (effective December 17, 2006).
[5] To attain this standard, the 3-year average of the fourth-highest daily maximum 8-hour average ozone concentrations measured at each monitor within an area over each year must not exceed 0.075 ppm. (effective May 27, 2008)
[6] (a) To attain this standard, the 3-year average of the fourth-highest daily maximum 8-hour average ozone concentrations measured at each monitor within an area over each year must not exceed 0.08 ppm.
(b) The 1997 standard – and the implementation rules for that standard – will remain in place for implementation purposes as EPA undertakes rulemaking to address the transition from the 1997 ozone standard to the 2008 ozone standard.
[7] (a) The standard is attained when the expected number of days per calendar year with maximum hourly average concentrations above 0.12 ppm is ≤1.
(b) As of June 15, 2005 EPA revoked the 1-hour ozone standard in all areas except the 8-hour ozone nonattainment Early Action Compact (EAC) Areas.
Source: USEPA website, Air and radiation. http://www.epa.gov/air/criteria.html (accessed 23 September 2008)

population and the fact that safe levels have to be inferred from uncertain epidemiological studies.

It is sometimes fruitful to regard the use of health damage as a criterion for determining pollution targets as an example of what may be called a 'modified efficiency target'. Our notion of efficiency to this point has been a comprehensive one; it involves maximising the difference between *all* the benefits of pollution and *all* the costs of pollution. But, sometimes, one particular kind of pollution cost (or damage) is regarded as being of such importance that pollution costs should be defined in terms of that cost alone. In this case we can imagine a revised or modified efficiency criterion in which the goal is to maximise the difference between all the benefits of pollution and this particular kind of pollution damage.

Policy makers sometimes appear to treat risks to human health in this way. So let us assume that policy makers operate by making risks to human health the only damage that counts (in setting targets). How would this affect pollution targets? The answer depends on the relationship between emissions and health risks. One possible relationship is that illustrated by the ⌋-shaped relationship in Figure 5.14. Total (and marginal) health risks are zero below the threshold, but at the threshold itself risks to human health become intolerably large. It is easy to see that the value of marginal benefits is irrelevant here. A modified efficiency criterion would, in effect, lead to the emissions target being set by the damage threshold alone. Target setting is simple in this case because of the strong discontinuity in human health risks. It is easy to see why an absolute maximum emission standard is appropriate.

But now suppose that marginal health damage is a rising and continuous function of emissions, as in

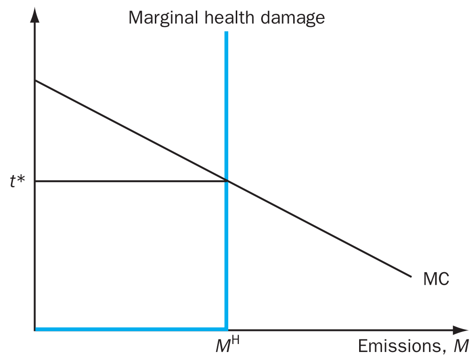

Figure 5.14 Setting targets according to an absolute health criterion

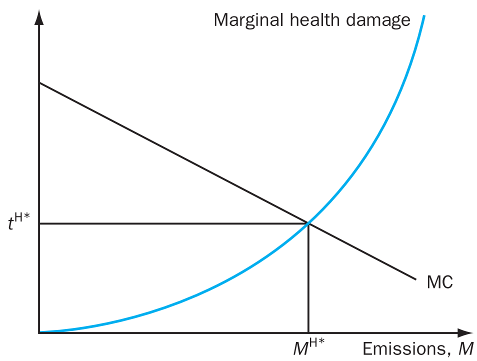

Figure 5.15 A 'modified efficiency-based' health standard

Figure 5.15. A trade-off now exists in which lower health risks can be obtained at the cost of some loss of pollution benefits (or, if you prefer lower health risks involve higher emission abatement costs). It is now clear that with such a trade-off, both benefits and costs matter. A 'modified efficiency target' would correspond to emissions level M^{H*}.[22]

Public preferences, sectional interests, and feasibility

Even ardent believers of the efficacy of democratic processes would be naïve if they did not recognise that public policy is influenced, sometimes very strongly, by the interplay of pressure groups and sectional interests. Moreover, in a world in which the perceived importance of international or global pollution problems is increasing, policy makers find themselves setting targets within a network of obligations to and pressures from various national governments, coalitions of such governments, and non-governmental organisations. Environmental policy making within this milieu is the subject of Chapter 9. This has been particularly evident in the area of international environmental agreements over such things as ozone depletion, acid rain and the greenhouse effect. National policy is also determined in the context of a network of pressures and influences. It is not surprising, therefore, that political feasibility plays a significant role, and that 'political economy' issues are key to environmental policy making in many areas such as global warming.

Public preferences are also about distributional concerns, and it is sometimes argued that environmental standards should be chosen so as to help achieve distributional goals. However, environmental standards are hardly the best, and certainly not the only, way of achieving distributional goals.

Summary

The concept of a pollution target

Unless government takes the view that pollution levels should be decided entirely by free market outcomes, a policy maker will, for any particular pollution problem, need to make a decision about how much of that pollution should be permitted. Sometimes – as in the case of greenhouse gas emissions – this target is announced explicitly in terms of a total amount of allowable emissions of the polluting

[22] Our analysis only attempts to scratch the surface of this set of issues. One set of additional matters that might warrant attention in practice (but which we do not cover here) arises from the complexities introduced by the existence of many health endpoints; that is, everything from minor coughs to premature mortality.

substance over some specified interval of time. Such choices logically precede any subsequent choices to be made about how that goal is to be achieved. Pollution problems may sometimes take different forms to those addressed in this chapter. One important kind is legacy effects, such as contaminated land. In that case, the choice concerns the rate at which the existing pollution stocks should be converted into harmless forms. Although we did not cover such legacy effects in this chapter, the techniques we have developed here can, with a little modification, be readily applied to such cases.

Different criteria can be used to determine pollution targets

The dominant approach taken by economists is to determine pollution targets on the presumption that the relevant objective criterion is economic efficiency. Much of this chapter has consisted of an exposition of that approach. However, economic efficiency is not the only relevant criterion for pollution target setting. Put another way, efficiency is only one of many criteria that policy makers might use – singly or jointly – in determining pollution targets. For many commentators, sustainability should be the principal criterion. In practice, much pollution target setting has been heavily determined by one principal damage route: impacts on human health and mortality. Several others were discussed in the chapter. Which criteria are important to policy makers will tend to reflect their policy objectives and the constraints under which they operate.

It is important to recognise that policy makers are likely to have multiple objectives. Efficiency matters, but it is not the only thing that matters. It is not surprising, therefore, that targets (or 'environmental standards' as they are sometimes called) are often chosen in practice on the basis of a mix of objectives. The mix may include health or safety considerations, equity, and perceptions of what is technically feasible (usually subject to some 'reasonable cost' qualification). In recent years, sustainability has taken its place as another stated goal of policy. As we show in Chapter 7, sustainability in conjunction with imperfect information and uncertainty may also point to some form of precautionary principle being incorporated in the set of objectives pursued by policy makers.

National and international policy is also determined in the context of a network of pressures and influences. Political feasibility, therefore, plays a significant role. This has been particularly important in the area of international environmental agreements over such things as ozone depletion, acid rain and the greenhouse effect, as we show in Chapter 9.

Alternative policy objectives usually imply different pollution targets

Different policy objectives usually imply different pollution targets. One particular comparison has received much attention in recent years: whether economic efficiency and sustainability are policy substitutes in the sense that the pursuit of the first implies the attainment of the latter. In general this will not be the case. As we showed earlier and will develop a little further later, efficiency based on the principle of consumer sovereignty is neither necessary nor sufficient for sustainability in the sense of resilience. So at this point suffice to say that there is no reason to believe that an efficient outcome will necessarily be a sustainable one. Equally, if health outcomes (or any other criterion) were used as a principal criterion in target setting, one would not in general expect that outcomes would be economically efficient. This should not be alarming in any way; there is nothing about efficiency which gives it any claim to superiority as an ultimate objective. Having said that, and other things being equal, there are good reasons for believing it is intrinsically desirable.

How pollution targets are constructed using an economic efficiency criterion

Understanding how pollution targets should be determined to achieve economically efficiency outcomes has been the main objective of this chapter. An economic efficiency criterion can be thought of as selecting pollution targets so as to maximise social net benefits, by taking full account of the trade-off between the costs and benefits of pollution (or, equivalent, the costs and benefits of pollution abatement). One should note, though, that in the context of limited or imperfect information, there may be immense difficulties in identifying economically efficient targets.[23] In that case, efficiency-based targets may be of theoretical interest only and have little practical significance. We examine this issue at some length in Chapter 7.

We do not expect pure (free and unregulated) market economies to deliver efficient outcomes in terms of pollution. Pollution tends to be an externality to the market process and as a result is not adequately reflected in private market decisions. Put another way, while firms would meet the costs of controlling or abating pollution, the benefits of abatement would not be received by firms (although they would by society). Hence, in considering pollution abatement, the control level that maximises net benefits to firms is different from the level that maximises social net benefits.

The difference between flow and stock pollutants

Being careful to make a proper distinction between flow and stock pollutants is important in environmental economics. Flow pollutants and stock pollutants differ in terms of the mechanisms by which damage is generated. This distinction has implications for the way in which targets are derived using an economic efficiency criterion. For stock pollutants, persistence implies that attention must be given to the accumulation (and decay) of pollutants over time, and so an intertemporal analysis is required. This is not necessary for the analysis of flow pollutants.

For long-lived stock pollutants, pollution targets are best thought of in terms of emissions paths over time. Efficient pollution paths will not in general imply the same level of control at all points in time. However, it is often useful to think of steady-state outcomes and to investigate what (constant) level of pollution control would be efficient in an equilibrium steady state.

The importance of the degree of mixing of a pollutant stock, and spatial differentiation

The degree of mixing of a pollutant stock has important policy implications both for emissions targets (as we discussed in this chapter) and for instruments (as we shall see in the following chapter.) Where a stock pollutant is not uniformly mixing, the spatial distribution of emissions sources becomes relevant. If targets are set in terms of pollutant concentrations, then the allowable emissions of any particular source will depend on its location.

The significance of non-convexities for environmental policy

Where emissions flows (or pollution stock) damage and/or benefit functions exhibit non-convexities, many of the standard analytical results may not be applicable. In particular, the equalization of marginal benefits with marginal costs may no longer be unique, and it may also be the case that an efficient outcome may not exist at any such equality. Hence, considerably more information may be required before policy makers can be confident that they have identified a target that would be fully efficient, as knowledge of the behaviour of costs and benefits over the whole domain of feasible values of emissions (or emissions abatement) is required, rather than just local information about relative magnitudes of costs and benefits in the neighbourhood of where the economy happens to be currently. As we shall see in the following chapter, the choice of instruments for achieving policy targets is also complicated by the presence of non-convexities as market-based instruments may not be able to support a competitive equilibrium.

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[23] Many of the problems posed by imperfect information also apply to targets set on the basis of sustainability, health, or indeed any other criterion. Nevertheless, they apply strongly to efficiency-based targets. However, as we shall see in Chapters 6 and 7, several of the alternative criteria can be interpreted as appropriate for target setting precisely when information is imperfect.

The distinction between first-best and second-best modes of analysis

This chapter has explained the difference between first-best and second-best modes of analysis. This distinction is central to most of the policy choices that governments must make in practice, as economies are typically characterized by multiple departures from the conditions required for first-best competitive equilibrium (and so efficient) allocation of resources. This is central to the debate about possible double dividends from tax reform packages. A second or double dividend might arise from a tax change that is intended to generate environmental improvements as some inefficiency elsewhere in the system might be mitigated by the selected reform package. It is, though, difficult to arrive at 'general results' in this area, as each tax reform package will have impacts that are context specific (and so analysis requires careful modelling of those specificities).

Further reading

Excellent and extensive presentations of the economics of pollution are to be found in Fisher (1981, chapters 5 and 6), Anderson (1985, 1991), Hartwick and Olewiler (1986, 1998) and Kolstad (2000). Baumol and Oates (1988) is a classic source in this area, although the analysis is formal and quite difficult. Cornes and Sandler (1996) provides a powerful theoretical underpinning in terms of the theory of public goods.

Tietenberg (1992) gives very extensive, descriptive coverage of several specific types of pollution. Other useful treatments which complement the discussion in this chapter are Dasgupta (1982, chapter 8), and two survey articles by Fisher and Peterson (1976) and Cropper and Oates (1992). Smith (1972) gives a mathematical presentation of the theory of waste accumulation. Several excellent articles can be found in the edited volume by Bromley (1995).

In this chapter we have taken a 'normative' approach to the setting of pollution targets, analysing what such targets should be in terms of some criterion of the public interest. An alternative literature considers targets in 'positive' terms, dealing with how targets are actually set. This approach focuses on the behaviour of interest groups, attempting to gain rents by manipulating government policy to their advantage. Good introductory accounts of this 'political economy' of regulation can be found in Goodstein (1995, 1999) and Kolstad (2000), particularly chapter 8. More advanced references are Laffont and Tirole (1993), which discusses theories of regulation, and Stigler (1971) and Peltzman (1976); these last two references are seminal works on the interest group theory of regulation.

Grubb *et al.* (1999) provides a very interesting account of greenhouse gas policy, focusing on technological responses to the Kyoto Protocol. Ulph (1997) considers the relationship between environmental policy and innovation. Porter (1991) articulates the argument that strict environmental policy may be a factor which stimulates the rate of technological innovation. The double dividend hypothesis is discussed by Bovenberg (1997). The collection of readings edited by Carraro and Siniscalco (1997) focuses on the application of game theory to environmental problems. This is a particularly useful tool in the analysis of international pollution problems, as we shall see in Chapter 9, but has interesting applications too for domestic pollution policy. One of the first studies about the difficulties in designing optimal taxes (and still an excellent read) is Rose-Ackerman (1973).

Some journals provide regular applications of the economic theory of pollution. Of particular interest are the *Journal of Environmental Economics and Management*, *Ambio, Environmental and Resource Economics*, *Land Economics*, *Ecological Modelling*, *Marine Pollution Bulletin*, *Ecological Economics* and *Natural Resources Journal*.

Discussion questions

1. 'Only the highest standards of environmental purity will do.' Discuss.
2. 'A clean environment is a public good whose benefits cannot be privately appropriated. Therefore private industry which is run for private gain will always be the enemy of a clean environment.' Examine this proposition.
3. Discuss the relevance and application of the concept of externalities in environmental economics.

Problems

1. Under which circumstances will the economically optimal level of pollution be zero? Under which circumstances will it be optimal to undertake zero pollution abatement?
2. We have seen that the efficient level of pollution is the one that minimises the sum of total abatement costs plus total damage costs. Refer now to Figure 5.3. Show that if pollution abatement takes place to the extent $\hat{M} - M_A$ the sum of total damage costs and total abatement costs is $C_1 + C_2 + C_3$. Prove that 'too little' abatement (relative to the optimal quantity) results in higher costs than $C_1 + C_2$.
3. Explain the concept of the 'efficient level of pollution'. What information is required in order to identify such an efficient quantity?
4. (a) Using equation 5.20 or 5.21, deduce the effect of (i) a decrease in α and (ii) an increase in r (*ceteris paribus*) on:
   (a) M^*
   (b) A^*
   (c) μ^*

   Note that you could answer this question analytically. Alternatively, you could explore the issue numerically using the Excel file *Stock1.xls* (found in the *Additional Materials* for Chapter 5).

   (b) A reader of the third edition of this text posed the following queries: 'I cannot understand, why in the case of alpha = 1, where there is no intertemporal problem, the optimality condition differs from the flow case? Also I do not understand, why in the case that there is no discounting the intertemporal solution equals the flow solution?' Provide the necessary intuition to answer these queries.
5. This problem illustrates how marginal analysis might give misleading results in the presence of non-convexity. It is based on an example from Goodstein (1995). Nitrogen oxides (NO_x), in combination with some volatile organic compounds and sunlight, can produce damaging lower-atmosphere ozone smog. Initially, the damage rises at an increasing rate with NO_x emissions. However, high levels of NO_x act as ozone inhibitors, and so beyond some critical level of emissions, higher levels of NO_x reduce ozone damage.
   (i) Sketch a marginal damage (MD) function that is consistent with these properties.
   (ii) Add to your diagram a conventionally shaped marginal benefits function (or marginal abatement cost function) that intersects the MD function in more than one place.
   (iii) By an appropriate choice of some initial level of emissions, demonstrate that the following rule may give misleading results. Rule: emissions should be increased (decreased) if a small increase in emissions increases (decreases) net benefits.

CHAPTER 6 Pollution control: instruments

Economists can only repeat, without quite understanding, what geologists, ecologists, public health experts, and others say about physical and physiological facts. Their craft is to perceive how economies and people in general will respond to those facts.

Dorfman (1985), p. 67

Learning objectives

After reading this chapter, the reader should understand

- how bargaining processes might bring about efficient resource allocations (and so might lead to the attainment of efficient pollution outcomes without regulatory intervention)
- the conditions which limit the likelihood of bargaining solutions to pollution problems being achieved
- the instruments available to attain a pollution target
- the mechanisms by which pollution instruments operate in attaining targets
- the comparative merits of alternative instruments
- the significance, in instrument choice, of whether a pollutant is uniformly mixing

Introduction

The previous chapter dealt with pollution targets. Here we consider how an environmental protection agency (EPA) could attain a predetermined pollution target by investigating the instruments that could be used.

In some circumstances no intervention would be required. Perhaps fortuitously, the prevailing level of pollution is not different from the target. Or intervention may be unnecessary because of the existence of voluntary bargaining. We show in Section 6.3 that bargaining between generators and victims of pollution could lead to an outcome in which the unregulated amount of pollution is equal to the pollution target. But we also show that such an outcome is unlikely for most important types of pollution problem. Where bargaining fails to reduce pollution to its targeted level, intervention of some form is called for.

This chapter is organised around three main themes. First, we describe the instruments that are available, and how each operates. Second, we consider whether there are particular circumstances – or particular types of pollution problems – which tend to favour the use of specific instruments. Of decisive importance is a matter raised in the previous chapter: whether or not the pollutant being targeted is uniformly mixing. Finally, we provide a comparative assessment of those instruments.

For the most part, our *analysis* will be quite general. That is, we will be thinking about instruments in the context of 'pollution problems' in general, rather than separately for air pollution, water pollution, soil contamination, and so on. However, the generality of the analysis will be limited in one important way. We will focus on pollution problems that are national (or sub-national) in scope, rather than on ones which are international. Control and regulation of *international* pollution problems will be addressed specifically in Chapter 9. The reason for segmenting the material in this way has nothing to do with the relative importance of different pollution problems. It is because dealing with international pollution issues brings another dimension into the picture: developing, coordinating and monitoring control across sovereign states. At this stage, we wish to keep this dimension out of our treatment.

Although the analysis in this chapter is general in its scope, the examples and applications deal with specific contexts and case studies, such as congestion charging in the UK. Several applications not covered in this chapter – specifically instruments for conserving biological diversity, mobile source (especially transport) pollution and its relation to climate change, and agricultural pollution (including reform of the Common Agricultural Policy to address environmental issues) – are examined in the Word files *Biodiversity*, *Transport* and *Agriculture* in the *Additional Materials* for Chapter 6. Carbon trading and the Kyoto Protocol and EU Emission Trading System are discussed in Chapter 9. Economic pressures causing deforestation are examined in Chapter 18.

6.1 Criteria for choice of pollution control instruments

There are many instruments available to an EPA charged with attaining some pollution target. How should it choose from among these? If attaining the target were all that mattered, instrument choice would be relatively simple. The best instrument would be the one that meets the target with greatest reliability. But the EPA is unlikely to have only this objective. Government typically has multiple objectives, and the terms of reference that policy makers impose on their agents will tend to reflect that diversity of objectives. Even where these terms of reference are not explicit, the network of influences and pressures within which the EPA operates will lead it to adopt multiple goals *de facto*.

Instrument choice can be envisaged in the following way. Each available instrument can be characterised by a set of attributes, relating to such things as impacts on income and wealth distribution, the structure of incentives generated, and the costs imposed in abating pollution. A score can be given to each instrument, dependent on how well its attributes match with the set of objectives sought by the EPA. (A hypothetical example of this is explored in Problem 1 at the end of this chapter.) This perspective is useful as it draws attention to what kinds of attributes a 'good' instrument might have. Table 6.1 lays out a set of criteria in terms of which the relative merits of instruments can be assessed.

The brief descriptions in the right-hand column of the table should be sufficient to convey what the various criteria mean. Fuller definitions and explanations of the first five items will be given later in the chapter. The remaining four all relate to decision making under conditions of limited information or uncertainty, and will be investigated in the next chapter. However, three observations about these criteria warrant mention now (but will be developed later).

Table 6.1 Criteria for selection of pollution control instruments

Criterion	Brief description
Cost-effectiveness	Does the instrument attain the target at least cost?
Long-run effects	Does the influence of the instrument strengthen, weaken or remain constant over time?
Dynamic efficiency	Does the instrument create continual incentives to improve products or production processes in pollution-reducing ways?
Ancillary benefits	Does the use of the instrument allow for a 'double dividend' to be achieved?
Equity	What implications does the use of an instrument have for the distribution of income or wealth?
Dependability	To what extent can the instrument be relied upon to achieve the target?
Flexibility	Is the instrument capable of being adapted quickly and cheaply as new information arises, as conditions change, or as targets are altered?
Costs of use under uncertainty	How large are the efficiency losses when the instrument is used with incorrect information?
Information requirements	How much information does the instrument require that the control authority possess, and what are the costs of acquiring it?

First, the use of any instrument is likely to involve conflicts or trade-offs between alternative criteria. Instrument choice will, therefore, depend on the relative weights attached to the criteria by the EPA. Second, it is likely that the weights (and so the choice of instrument) will vary over different types of pollution. For example, where a dangerous and persistent toxin such as the synthetic pesticide DDT or heavy metals such as lead or mercury is concerned, the EPA may regard cost efficiency as being of low importance relative to the long-run effect of the chosen instrument. Third, no single instrument is best for dealing with all types of pollution in all circumstances. We shall see in the next chapter that this is true *a fortiori* where instrument choice takes place under conditions of uncertainty.

One particular criterion – cost efficiency – has received so much attention in the environmental economics literature that it warrants special attention now.

6.2 Cost efficiency and cost-effective pollution abatement instruments

Suppose a list is available of all instruments which are capable of achieving some predetermined pollution abatement target.[1] If one particular instrument can attain that target at lower real cost than any other can then that instrument is cost-effective.[2] Cost-effectiveness is clearly a desirable attribute of an instrument. Using a cost-effective instrument involves allocating the smallest amount of resources to pollution control, conditional on a given target being achieved. It has the minimum opportunity cost. Hence, the use of cost-effective instruments is a prerequisite for achieving an economically efficient allocation of resources.[3]

Let us explore some ramifications of the cost-effectiveness criterion. There will (usually) be many sources of an emission, and so many potential abaters. This raises the question of how the overall target should be shared among the sources. The principle of cost efficiency provides a very clear answer: a necessary condition for abatement at least cost is that the marginal cost of abatement be equalised over all abaters. This result is known as the least-cost theorem of pollution control. It is derived algebraically in the first part of Appendix 6.1. You will find it useful to read that now.

The intuition behind this result is easily found. Consider a situation in which marginal abatement costs were not equalised. For example, suppose that at present abatement levels two firms, A and B, have marginal abatement costs of 60 and 100 respectively. Clearly if B did one unit less abatement and A did one more (so that total abatement is unchanged) there would be a cost reduction of 40. Cost savings will accrue for further switches in abatement effort from B to A as long as it is more expensive for B to abate pollution at the margin than it is for A.

Let us examine these ideas a little further.[4] Suppose government wishes to reduce the total emission of a particular pollutant from the current, uncontrolled, level $\hat{M}$ (say, 90 units per period) to a target level M^* (say, 50 units). This implies that the abatement target is 40 units of emission per period. Emissions arise from the activities of two firms, A and B. Firm A currently emits 40 units and B 50 units.

The following notation is used. The subscript i indexes one firm (so here i = A or B). M_i is the actual level of the ith firm's emissions, which will depend on what control regime is in place. Two particular levels are of special interest. $\hat{M}_i$ is the profit-maximising level of emissions by firm i in the absence of any controls set by government and in the absence of any pollution charges. M_i^* is an emission ceiling (upper

[1] You will notice that we refer here to a pollution reduction (or abatement) target, rather than to a target level of pollution itself. This conforms to conventional usage in the literature on instruments. In this chapter, the context should make it clear whether the target being referred to relates to pollution or pollution abatement.

[2] Strictly speaking, an instrument is cost-effective if its real resource cost is no greater than that of any other instrument available. This means that a cost-effective instrument may not be unique. For example, suppose that two instruments each incur costs of £10m to bring sulphur dioxide pollution down to some target level, while all others cost more than £10m. Then those two instruments are cost-effective.

[3] It is this which explains why the cost-effectiveness criterion has figured so prominently in the economics literature.

[4] The following problem is replicated in the Excel workbook *Leastcost.xls*, found in the *Additional Materials* for Chapter 6.

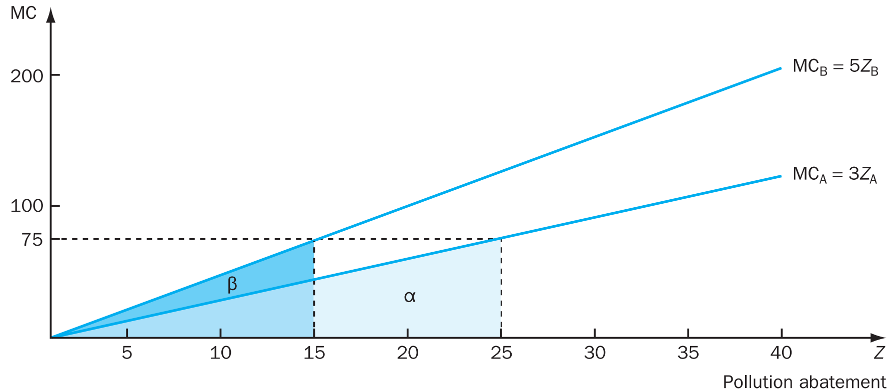


Figure 6.1 Marginal abatement cost functions for the two firms

limit) set for the firm by the EPA. The quantity of pollution abatement by the ith firm is Z_i, given by $Z_i = \hat{M}_i - M_i^*$. Hence we assume that whenever an emissions regulation is in operation the amount of emissions the firm actually produces is that set by the EPA. C_i is the total abatement cost of the ith firm.

Suppose that the total abatement cost functions of the two firms are $C_A = 100 + 1.5Z_A^2$ and $C_B = 100 + 2.5Z_B^2$. Therefore, the marginal abatement cost functions are $MC_A = 3Z_A$ and $MC_B = 5Z_B$. These are sketched in Figure 6.1. The least-cost solution is obtained by finding levels of Z_A and Z_B which add up to the overall abatement target $Z = 40$ *and* which satisfy the least-cost condition that $MC_A = MC_B$. This gives the answer $Z_A = 25$ and $Z_B = 15$. Figure 6.1 shows this least-cost solution. At those respective abatement levels both firms have marginal abatement costs of 75. Minimised total abatement costs (1700) can be read from the diagram. The darker shaded area denoted β shows B's total abatement costs (662.5), while the lighter area denoted α represents A's total abatement costs (1037.5).

To verify this result, you could use the Lagrange multiplier technique, obtain the necessary first-order conditions, and solve these for the two firms' abatement levels. This was explained in the appendix to Chapter 3, where this problem – albeit with different numbers – was solved to show how the technique works. A convenient alternative, taking only a couple of minutes, is to use Excel's Solver routine to do this task for us. The mechanics of doing so are given in *Leastcost.xls* (in *Additional Materials*, *Chapter 6*) and you are recommended to study this Excel workbook now.

It is instructive to compare this solution with two others. First, one might think that as firm A has a lower marginal abatement cost schedule than B it should undertake all 40 units of abatement. It is easy to verify that this results in higher costs (2500) than those found in the least-cost solution (1700). Second, an equity argument might be invoked to justify sharing the abatement burden equally between the two firms. But it is easy to show (for example by looking at *Sheet1* of *Leastcost.xls*) that this also leads to higher costs (1800 in fact). If the regulator wanted such an equitable outcome, it would come at an additional real cost to the economy of 100 units (1800 − 1700). Note that the greater the difference in the firms' abatement cost functions, the greater would be the cost penalty from not pursuing the least-cost outcome. (See Problem 2.) Some important conclusions emerge from this analysis:

- A least-cost control regime implies that the marginal cost of abatement is equalised over all firms undertaking pollution control.
- A least-cost solution will in general not involve equal abatement effort by all polluters.
- Where abatement costs differ, cost efficiency implies that relatively low-cost abaters will undertake most of the total abatement effort, but not all of it.

We shall use these results later in this chapter to establish whether particular kinds of pollution control instrument are cost-effective.

6.3 Instruments for achieving pollution abatement targets

In this section, we describe and explain the instruments available for pollution control. For convenience, the most common are listed in Table 6.2. Our emphasis is on the method of operation of each instrument and whether the instrument is cost-efficient. A more complete examination of the relative advantages of the instruments is left until later in the chapter.

6.3.1 Approaches which facilitate voluntary, decentralised internalisation of externalities

One approach to achieving emissions, or other environmental policy, targets is to improve existing social or institutional arrangements that facilitate environmental damage-reducing voluntary decentralised behaviour. Two variants of this approach are:

1. improve the effectiveness of property rights regimes in bringing about socially efficient allocations of resources;
2. encourage greater social responsibility in making choices and taking decisions.

Each of these two variants shares the characteristic of potentially preventing the emergence of externalities, or internalising externalities which have arisen. In doing so, it is possible that decentralised behaviour by consumers and producers may generate efficient outcomes and so obviate the need for regulatory intervention. To the extent that this is feasible (and cost-effective), the need to use other pollution control instruments would be correspondingly reduced.

6.3.1.1 Bargaining solutions and the limitations on bargaining solutions to environmental problems

In a classic paper, Ronald Coase (1960) explored the connection between property rights and the likelihood of efficient *bargaining* solutions to inefficient allocations of resources. This literature was subsequently greatly expanded, particularly by writers in what became known as the 'New Institutional Economics' school.[5] Coase proposed that a necessary condition for bargaining between agents to bring about efficient resource allocation is the existence of a well-defined and enforceable allocation of property rights.[6]

We shall investigate the policy implications of this proposition shortly. First of all, let us review the way in which bargaining may be able to internalise externalities and so achieve efficient outcomes. In Chapter 4, we considered an example in which the noise generated by a musician disturbed a neighbour. We showed how bargaining between those two parties could generate an efficient quantity of music playing. However, our discussion also demonstrated that efficient bargaining outcomes are often hard to obtain, and are sometimes impossible. These limitations are particularly likely for many kinds of environmental problem. Why this should be so?

1. First, as Coase argued, the likelihood of bargaining taking place is at best low unless well-defined and enforceable property rights exist. For many environmental resources, well-defined and enforceable property rights do not exist. An important example is that in which the environmental resource is an open-access resource in which exclusion is impossible except at very high, and possibly prohibitive, cost.
2. Second, bargaining solutions require that the expected gains from bargaining are larger than the expected costs of carrying out that bargaining. Thus, bargaining is facilitated by the existence of a relatively small number of

[5] The Further Reading at the end of this chapter lists some of the seminal works relating to property-rights regimes and their impacts on productive and allocative efficiency in the New Institutional Economics literature.

[6] Coase also showed that efficient bargaining may be hindered by the presence of non-trivial transactions costs.

Table 6.2 Classification of pollution control instruments

Instrument	Description	Examples (Notes 1, 2)
Institutional approaches to facilitate internalisation of externalities		
Facilitation of bargaining	Cost of, or impediments to, bargaining are reduced	Polluter information placed in the public domain
Specification of liability	Codification of liability for environmental damage	Used to compensate for respiratory damage in Japan
Development of social responsibility	Education and socialisation programmes promoting 'citizenship'	Energy-conservation media campaigns; Environmental labelling
Command and control instruments		
Input controls over quantity and/or mix of inputs	Requirements to use particular inputs, or prohibitions/restrictions on use of others	Bans on use of toxic cleansing agents
Technology controls	Requirements to use particular methods or standards	Requirement to install catalytic converters in exhausts. BAT, BATNEEC (3)
Output quotas or prohibitions	Non-transferable ceilings on product outputs	Ban on use of DDT Vehicle quotas. Effluent discharge licences
Emissions licences	Non-transferable ceilings on emission quantities	
Location controls (zoning, planning controls, relocation)	Regulations relating to admissible location of activities	Heavy industry zoning regulations
Economic incentive (market-based) instruments		
Emissions charges/taxes	Direct charges based on quantity and/or quality of a pollutant	Air pollution charges (e.g. on NO_X and SO_2) Carbon/energy taxes Water effluent charges Noise pollution charges Fertiliser and pesticide taxes
User charges/fees/natural resource taxes	Payment for cost of collective services (charges), or for use of a natural resource (fees or resource taxes)	User charges on municipal waste collection, treatment or disposal Hazardous waste charges Wastewater charges Aircraft noise charges Water extraction charges Congestion pricing
Product charges/taxes	Applied to polluting products	Taxes or charges on vehicle tyres, nuclear waste, plastic bags, other disposables
Emissions abatement and resource management subsidies	Financial payments designed to reduce damaging emissions or conserve scarce resources	Subsidy for energy generated from waste Grants to ecological farming
Marketable (transferable, marketable) emissions permits	Two systems: those based on emissions reduction credits (ERCs) or cap-and-trade	CO_2 emissions from power plants
Deposit-refund systems	A fully or partially reimbursable payment incurred at purchase of a product	Refillable plastic bottles Charges on one-way beer and soft-drink bottles
Non-compliance fees	Payments made by polluters or resource users for non-compliance, usually proportional to damage or to profit gains	Sea dumping of oil from ships
Performance bonds	A deposit paid, repayable on achieving compliance	Australia: mine sites US: open pits
Liability payments	Payments in compensation for damage	Restoration of sites polluted by illegal dumping

1. Many of the examples in the table are drawn from OECD (1999) and EPA (1999). These references were available online at time of writing from OECD and USEPA websites. They provide extensive accounts of incentive-based environmental controls used in OECD countries. Current web addresses are available via the version of this table on the Companion Website.
2. Particular cases are mentioned purely as examples. Listings are not exhaustive. A fuller, more complete, and periodically updated version of this table, with specific country examples, is found in the Additional Materials section on the Companion Website.
3. BAT is 'Best Available Technology'; BATNEEC is 'Best Available Technology, Not Entailing Excessive Cost'.

affected parties, and by all such parties being easily identifiable. Again, many environmental problems fail to satisfy either of those properties. Typically, environmental degradation affects many people and in many cases, as with vehicle pollution, is attributable to a large number of sources. It is often difficult to identify all affected parties, and the transactions costs associated with undertaking a bargaining exercise can be enormous. Hence where the number of affected individuals is large, the scope for efficient bargaining behaviour is restricted.

3. Another pertinent issue relates to the possibility of intertemporal bargaining, including bargaining between current and future generations. Often, environmental externalities cut across generations – our behaviour today imposes externalities on future persons. While bargaining between affected individuals at one point in time seems feasible, it is difficult to imagine that this could happen between representatives of the present generation and those not yet living. One would not, therefore, expect that bargaining between directly affected individuals and firms would offer much prospect of bringing about an efficient response to global climate change, involving as it does many generations.

What do these observations imply about the role for government? If bargaining does offer the prospect of substantial efficiency gains, then government should facilitate it wherever that is cost-effective. It could do so by clearly defining and explicitly allocating property rights where that is practicable (and ethically acceptable). Of course, where environmental problems spill over national boundaries, as in the case of biodiversity decline or greenhouse gas emissions, further complications arise. In the absence of international governmental institutions with full sovereignty, internationally agreed and enforceable property rights over environmental resources often do not exist. Indeed, a part of what takes place in negotiation processes leading to what are hoped will be international environmental agreements can be interpreted as attempts to create agreed and effective property rights. For example, the Convention on Biological Diversity has grappled with the issue of how to create 'host country' property rights in biological materials. Equally, much of the rancour that is associated with discussions about limitations of greenhouse gas emissions is effectively about property rights in the atmosphere. These are two examples of where international bargaining has the potential to generate massive collective benefits, but where bargaining between sovereign states in the absence of an internationally agreed property rights regime makes the realisation of those gains extremely difficult. As international policy cooperation about environmental problems is the subject of a separate chapter (Chapter 9), we shall postpone further consideration of this matter until then.

Returning to the single nation context, government might also seek to develop and sustain an institutional structure that maximises the scope for bargaining behaviour.[7] Where the number of potential bargainers is very small, one can readily find examples, such as marriage counselling services, arbitration mechanisms for employment disputes, and plea bargaining procedures. But where the number of affected individuals is large, as is the case for most environmental problems, one should not be surprised that they are less common.

Gains may also derive from government's taking some responsibility for environmental monitoring so as to identify pollution producers and recipients, and disclosing information from this to affected parties. Finally, access to the judicial system should be easy and cheap, and also equitable as between different classes of parties. This will also facilitate use of the liability principle that we shall discuss in the next section.

One has to conclude that the limitations to bargaining that we have described do appear to be very

[7] Elinor Ostrom (1990) has shown that in many societies bargaining solutions to resolve disputes are often embedded in long-standing cultural traditions and social norms, and collective choice mechanisms operating within these frameworks. These social structures can be of great efficacy and can bring about efficient resources allocations even in the context of common-property (as opposed to private-property) regimes. However, increased complexity of social and economic systems, along with greater geographical and social mobility, tends to weaken those traditions and norms.

substantial, and it would be inappropriate to place too much reliance on such a mechanism as far as environmental pollution problems are concerned.[8]

6.3.1.2 Liability

The role that may be played by the judicial system in helping to bring about efficient outcomes has been implicit in our discussion of bargaining. But that role can be taken a step further. Liability can be used as a means of dealing with environmental hazards. Suppose that a general legal principle is established which makes any person or organisation liable for the adverse external effects of their actions. Thus if you harm another person, you may be required to compensate that person for the damage done.[9] In effect, property rights are then vested in the party adversely affected by the action which generates the harm.

It is useful to think about liability in the context of risky activities. An appropriate public policy response to activities that are beneficial but also generate positive risks of harmful outcomes would be one that induces precautionary, risk-reducing, behaviour by those who undertake risky activity. But what level of precaution is warranted? Figures 6.2 and 6.3 help us answer this question. We use Q to denote the amount of precaution undertaken by the potential injurer. In Figure 6.2, the vertical intercept of the chart shows the *expected value* of damage that would occur if an organisation takes no mitigating precautionary actions. The expected value of damage is the product of two things: the probability of a harmful outcome happening, p; and the value of any damage actually done, D. For example if an accident has a 1 in 100 chance of happening ($p = 0.01$) over some specified interval of time and such an accident would cause damage to the value £10,000 (D = £10,000), then the expected value of the damage is $p \times D$ = £100. However, by taking precautionary actions – that reduce p, reduce D, or both – the expected value of damage falls. The greater the amount of precaution, the larger is this reduction in expected value of damage.

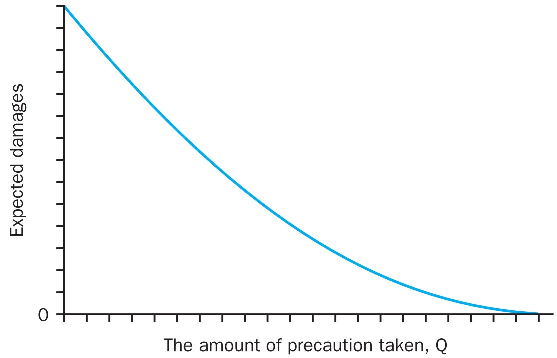


Figure 6.2 Expected damages, reducing as the amount of precaution taken, *Q*, increases

Now look at Figure 6.3. The top graphic shows the total social benefit (B) and the total cost (C) associated with each amount of precautionary behaviour. Social benefits arise from the *expected value* of damage that is **averted** by precautionary, risk-reducing behaviour. We obtain the total social benefit function, B, as the negative of the expected value of damages. Costs, C, arise from additional resources being devoted to precaution, either intended to reduce the likelihood of an accident actually happening, p, or to reduce the magnitude of harm done should the accident happen, D. We assume that both functions are well behaved with B increasing as Q rises, but at a decreasing rate, while C is increasing at an increasing rate.

The level of precaution applied is socially efficient when the net benefit ($B - C$) from precaution is maximised. This is shown at $Q = Q^*$ in Figure 6.3. Note that the slopes of the B and C functions are equalised at this point. Hence, as shown in the lower half of the graphic, the *marginal* quantities are equated. That is, the marginal benefit of precaution and the marginal cost of precaution are equal at the socially efficient level of precaution, Q^*. That is, $MB(Q^*) = MC(Q^*)$.

What we now seek is an incentive mechanism that induces the potential injurer to undertake the socially efficient level of precaution, Q^*. Liability

[8] However, when it comes to dealing with pollution, or other environmental, problems that spill over national boundaries, the absence of supra-national sovereign institutions means that there is often little or no alternative to bargaining solutions.

[9] We write 'may be required' rather than 'will be required' for a reason that will become apparent shortly.

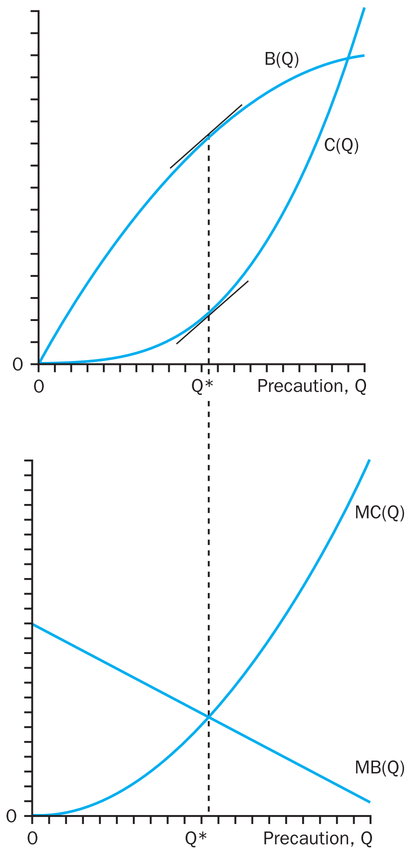


Figure 6.3 The socially efficient level of precautionary behaviour

for damage might be one such mechanism. Let us consider two possible versions of the liability principle (there are many others) and investigate whether they generate the appropriate incentive.

1. Strict liability — If an accident occurs, the injurer pays full compensation to the victim.
2. Negligence liability — If an accident occurs, the injurer pays full compensation to the victim only if the injurer were not, prior to the accident, undertaking the efficient level of precaution. If it were, then the injurer is not required to pay any compensation.

Will either of these liability rules create incentives that induce any organisation that undertakes risky behaviour to undertake the socially efficient amount of precaution, Q^*? To answer this question, first note that under either of these two liability principles, if compensation is actually paid the amount of that compensation is the actual value of the damage done, D, not the expected value of the damage ($p \times D$). However, *ex ante* (that is, prior to having any knowledge about whether an accident will actually occur in some following interval of time) a potential injurer expects that it will make compensation payments to the amount $p \times D$ (as there is a probability p that a payment of D will have to be made).

Figure 6.3 shows it to be worthwhile for a potential injurer to undertake additional precaution as long as the marginal cost of additional precaution, MC(Q), is less than the marginal benefit of additional precaution, MB(Q). But MB(Q) is the change in the expected value of damage when a little extra precaution is taken. So it is evident that the strict liability principle will generate the correct incentive, as following this rule would lead to precaution being increased until the point where $Q = Q^*$.

Although it is not so obvious, the negligence liability principle also creates an incentive on organisations to set $Q = Q^*$. To demonstrate that this is so, one could ask which level of precaution minimises the sum of costs incurred in reducing risk and expected compensation payments. Doing this shows that Q^* minimises that sum of costs. We shall not go through the reasoning here, but instead will leave this as an exercise for the reader. (See Problem 11, and its answer on the Companion Website).

We remarked earlier that using liability as a pollution control instrument is akin to creating property rights. The Coase theorem (Coase, 1960) suggests that, under particular circumstances, the same efficient outcome will be achieved irrespective of the manner in which property rights are initially allocated, but that the distributional consequences can be very different. We see an example of this in the two different forms of the liability principle. Strict liability and negligence liability both produce the same efficient outcome, $Q = Q^*$. However, their distributional impacts are different. Strict liability is equivalent to granting property rights to potential victims of damage. Those victims are entitled to full compensation for damages done to them, irrespective of the level of *ex ante* precaution taken by the injuring party. Negligence liability, in contrast, grants property rights to the potential injurer, provided that it has undertaken the socially efficient level of precaution. For in that case, the compensation payments to be made are zero. Discussion Question 1 invites you to consider further the relative merits of the two forms of liability principle in the light of these considerations.

But as with all matters relating to property rights, things are more complicated where the damage or harm done is a public rather than a private good. Where harm is a public good, use of liability as way of making the polluter pay is not usually feasible. In that case, it may be efficient (and perhaps also ethically attractive) for the EPA to act as an agent of the public interest, enforcing the liability principle on behalf of affected parties. An interesting question is whether any damages obtained in this way should be returned to individuals as compensation. We explore this matter in Discussion Question 2.

Use of the liability principle also faces additional problems. One difficulty arises where damage only becomes apparent a long time after the relevant pollutants were discharged. Tracking down those who are liable may be a substantial undertaking, and those responsible – individuals or firms – may no longer exist. Related to this is a wider class of pollution problems in which actions undertaken in earlier times, often over decades or even centuries, leave a legacy of polluted water, land, or biological resources. Even if one could identify the polluting culprits and apportion blame appropriately, it is not clear whether an *ex post* liability should be imposed. The Further Reading section at the end of this chapter points you to some of the literature on this topic.

Finally, the way in which liability should operate is complicated when production activities are linked in a complex way along a supply chain. In many cases, pollution or other environmental damage is incurred by activities of final product consumers, often at the point of disposal of the final good, and perhaps a substantial time after production its original supply chain. An interesting development is the process of establishing legal liability throughout the life cycle of a product, using the principle that producers are responsible for damage from 'cradle to grave'. An example of this principle is illustrated in Box 6.1.

Box 6.1 The liability principle in action

The use of liability payment schemes is widespread. In many countries, common law systems have long provided some opportunity for legal redress against damages, but the operation of this has typically been uncertain and costly, and compensation has been determined on somewhat ad hoc grounds. During the latter part of the twentieth century, many governments began to augment common law provision by moving towards formal liability schemes for specified categories of pollutant (such as titanium dioxide pollution in Quebec; noise pollution in Germany; and hazardous waste in the USA). A more recent trend has been for countries to codify general (i.e. non pollutant-specific) liability schemes. In some cases, such as in Finland, compulsory environmental damage insurance has been required for large polluters.

Since the 1970s, Japanese courts have developed an extensive liability case law, relating primarily to waste, air and water pollution. Japanese businesses contribute to a compensation fund. Until 1988, persons with bronchial asthma and other respiratory diseases were entitled to compensation from the fund without judicial procedure. After 1988, new sufferers were no longer entitled to automatic compensation, as air pollution was no longer unequivocally accepted as the principal contributory factor to respiratory illnesses.

An interesting example of the liability for damage principle can be found in the regulations relating to hazardous waste disposal in the USA. Under the terms of the Resource Conservation and Recovery Act, a 'cradle-to-grave' tracking and liability principle has been adopted. The *Superfund* concerns abandoned waste dumps. The fund is built up from various sources, including damages settlements. The principle of 'strict, joint and several liability' establishes a special form of retrospective liability, in which parties that have dumped waste (legally or illegally) can be sued for the whole costs of clean-up, even though they were only partial

Box 6.1 *continued*

contributors to the dump. The sued party may then attempt to identify others responsible to recover some of the damages. Moreover, liability lies with the generators of waste as well as those who subsequently reprocess or dispose along the waste cycle.

A European Union (EU) Directive (2004/35/EC) seeks to prevent and remedy environmental damage by making operators financially liable for threats of or actual damage. The directive covers what is deemed to be 'more serious' damage to: habitats and species protected by European Commission law; species or habitat on a site of special scientific interest for which the site has been notified; water resources; and land contamination which presents a threat to human health. EU Member States are required to introduce procedures at national level to implement the directive.

When damage is threatened the relevant EPA can impose precautionary obligations on operators of potentially damaging activities which cause or threaten to cause environmental damage. (To what extent those obligations will correspond to what this chapter has called 'socially-efficient precaution' remains to be seen.) For actual damage, the Directive specifies two types of liability: fault-based liability (what we called 'negligence liability') in respect of environmental damage to protected species and natural habitats from all other occupational activities, and strict liability in respect of environmental damage, caused by a specified range of 'occupational activities'.

Individual member countries all have some pre-existing liability regimes. In the UK, for example, regulations which provide for the remediation of environmental damage include the Environmental Protection Act 1990, the Water Resources Act 1991, the Wildlife and Countryside Act 1981 and the Control of Major Accident Hazards Regulations 1999. These can require damage to be put right by those responsible for it, or allow the controlling authority to put the damage right themselves and then recover the costs afterwards from those responsible. The new Regulations arising from the EU Directive will supplement existing environmental protection legislation; those earlier pieces of legislation will still apply, and to the extent that they impose additional obligations to those in these Regulations, will still need to be complied with.

Sources: OECD (1999); UK DEFRA website, via http://www.defra.gov.uk/environment/liability/, accessed 26 March 2009; USEPA website (Superfund pages), via http://www.epa.gov/superfund/, accessed 26 March 2009

6.3.1.3 Development of social responsibility

Pollution problems happen, in the final analysis, because of self-interested but uncoordinated behaviour. Encouraging people – either as individuals or in their roles within organisations – to behave as socially responsible citizens can help to attain environmental goals. Clearly, any government has limited influence over the cultural context of human behaviour. But it would be wrong to ignore the opportunities that exist for using educational institutions and the mass communications media to help achieve specific targets and to promote ethical behaviour.

The evidence that individuals do not exclusively act in a narrowly utilitarian way suggests that this objective may be more than just wishful thinking. Among the very many examples that could be cited are support for green parties and the increasing importance being given to environmental issues by voters, the success of some ethical investment funds, and our willingness to support charities. Perhaps the strongest evidence is to be found in our family and social lives, where much of what we think and do has a social – rather than purely self-interested – basis. Given this, 'cultural' instruments that promote 'social responsibility' may be powerful ways of achieving general environmental goals. One particular policy mechanism which could be said to be in the category is environmental labelling, used in virtually all industrialised economies and in many developing countries. This has been credited with reducing VOC (volatile organic compound) emissions in Germany, and with increasing paper recycling in Korea (EPA, 1999).

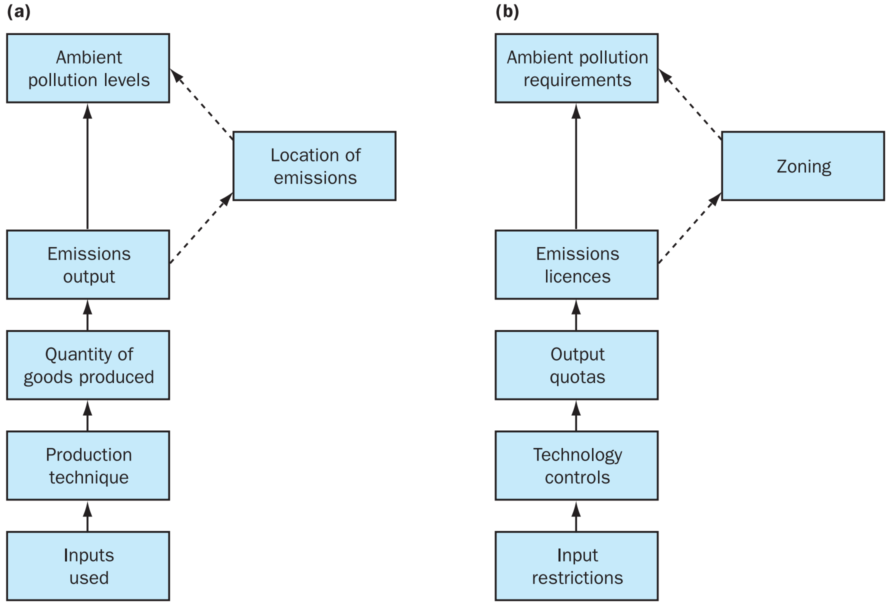


Figure 6.4 A classification of command and control instruments

6.3.2 Command and control instruments

The dominant method of reducing pollution in most countries has been the use of direct controls over polluters. This set of controls is commonly known as *command and control* instruments. Figure 6.4 provides a schema by which these instruments can be classified. The first panel (Figure 6.4a) represents the various relationships that link production to pollution levels. Emissions are by-products in the production of intended final output. The amount (and type) of emissions will depend on which goods are being produced, and in what quantities. It will also depend on the production techniques being employed, and on the amount (and mix) of inputs being used. For uniformly-mixing pollutants (UMPs), pollution levels will depend only on total emissions levels. In the case of non-uniformly-mixing pollutants (indicated in the diagram by the dotted lines in the branch to the right), the spatial distribution of ambient pollution levels will also depend on the location of emission sources.

Command and control instruments can be designed to intervene at any of these stages. So, as the second panel (Figure 6.4b) illustrates, regulations may apply to outputs of emissions themselves, to the quantity of final production, to production techniques used, or to the level and/or mix of productive inputs. For non-UMPs, controls may also apply to location of emission sources.

In general, there should be advantages in directing the controls at points closest (in this sequence of linkages) to what is ultimately being targeted: that is, ambient pollution levels. This allows polluters the most flexibility in how a pollution reduction is to be achieved. But it may not be feasible or cost-effective to set regulations in that way.

There is huge variation from place to place in the detail of regulatory systems. It would be of little use – and probably impossible – to list the plethora of command and control regulations. Our coverage of command and control instruments will be limited, therefore, to some general comments on the main categories shown in Figure 6.4b, together with some illustrative examples in boxes. For further detail, the reader should visit the text's Companion Website, which provides links to many sites that provide regularly updated accounts of regulatory regimes in various countries.

We next describe the three most commonly used types of command and control instrument, and in doing so make some preliminary remarks about the likely cost-efficiency of each. A more complete appraisal of the relative merits of each class of instrument covered in this chapter, including a discussion of some empirical evidence, will be left until Section 6.6.

6.3.2.1 Non-transferable emissions licences

Suppose that the EPA is committed to attaining some overall emissions target for a particular pollutant. It creates licences (also known as permits or quotas) for that total allowable quantity. After adopting some criterion for apportioning licences among the individual sources, the EPA distributes licences to emissions sources. These licenses are *non-transferable*; that is, the licences cannot be transferred (exchanged) between firms. Therefore, each firm's initial allocation of pollution licences sets the maximum amount of emissions that it is allowed.[10] Successful operation of licence schemes is unlikely if polluters believe their actions are not observed, or if the penalties on polluters not meeting licence restrictions are low relative to the cost of abatement. Licence schemes will have to be supported, therefore, by monitoring systems and by sufficiently harsh penalties for non-compliance.

Under special conditions, the use of such emissions licences will achieve an overall target at least cost (that is, be cost-efficient). But it is highly unlikely that these conditions would be satisfied. We know (see the first part of Appendix 6.1) that cost-efficiency requires the marginal cost of emissions abatement to be equal over all abaters. If the EPA knew each polluter's abatement cost function, it could calculate which level of emissions of each firm (and so which number of licences for each firm) would generate this equality *and* meet the overall target.

It is very unlikely that the EPA would possess, or could acquire, sufficient information to set standards for each polluter in this way. The costs of collecting that information could be prohibitive, and may outweigh the potential efficiency gains arising from intervention. Moreover, there is a problem of information asymmetries; those who possess the necessary information about abatement costs at the firm level (the polluters) do not have incentives to provide it in unbiased form to those who do not have it (the regulator).[11] We examine these incentives in a little more detail in Section 6.6. A system of long-term relationships between regulator and regulated may partially overcome these asymmetries, but might bring other problems (such as regulatory capture – to be defined and explained in Chapter 7). Given all this, it seems likely that arbitrary methods will be used to allocate licences, and so the controls will not be cost-efficient.

6.3.2.2 Instruments which impose minimum technology requirements

A second category of command and control instrument consists of regulations which specify required characteristics of production processes or capital equipment used. In other words, minimum technology requirements are imposed upon potential polluters. Examples of this approach have been variously known as *best practicable means* (BPM), *best available technology* (BAT) and *best available technology not entailing excessive cost* (BATNEEC).

In some variants of this approach, specific techniques are mandated, such as requirements to use flue-gas desulphurisation equipment in power generation, designation of minimum stack heights, the installation of catalytic converters in vehicle exhaust systems, and maximum permitted lead content in engine fuels. In other variants, production must employ the (technically) best technique available (sometimes subject to a reasonable cost qualification). The specific technique adopted is sometimes negotiated between the EPA and the regulated parties on an individual basis. Some further information on technology controls is given in Box 6.2.

Much the same comments about cost-effectiveness can be made for technology controls as for licences. They are usually not cost-efficient, because the

[10] We use the term 'licence' to denote non-transferable emissions quotas. Later in the chapter, transferable quotas will be discussed. To avoid confusion, we call these 'permits'.

[11] Another possibility is that firms themselves may also be unaware of their abatement costs.

Box 6.2 Required technology controls

Regulations mandating the use of particular technologies are common forms of pollution control instrument in Europe, North America and the other OECD countries. In the UK, a criterion underlying required technology standards has been 'best practicable means'. The adjective *practicable* has never been given a precise legal definition, but the 1956 Clean Air Act stated that

> Practicable means reasonably practicable having regard, amongst other things, to local conditions and circumstances, to the financial implications and the current state of technology.

Despite an element of tautology in this statement, it can be interpreted as meaning that a practicable control technology should be technologically effective, subject to the constraint that it is not excessively costly. In recent years, the cost qualification has been given greater priority, and has been enshrined in the principle of BATNEEC: the best available technology not entailing excessive cost.

The European Union Water Framework Directive, which came into force in December 2000, imposes the (very strong) requirement that every inland and coastal waters river or other water body in each EU member state must attain 'good chemical and ecological status' by 2015. For many water bodies in Europe, the only way that this could be achieved is by ensuring that all water and sewage infrastructure is of best practicable technology. However, the costs of attaining this standard for some heavily polluted water bodies would be prohibitive. Therefore, member states are allowed to seek permission to 'derogate' (i.e. defer, or in extreme cases, not pursue) attainment of the good water status standard if evidence can be given to show that the costs of achieving that standard would be excessive.

The use of a 'not at excessive cost' qualifier on required technology standards puts this instrument closer to the kind advocated by economists, as the 'excessive cost' condition implies a quasi-cost–benefit calculation in the administration of the instrument. However, while the cost of control is often measured in money terms (for example, the additional money cost of one technique over another), the benefits are not usually measured in money terms; instead, benefits are seen in terms of reduced probabilities of death or serious damage to health. In this sense, although some balancing of costs against benefits does often take place, the approach being used is not 'cost–benefit analysis' in the sense we shall be using in Chapter 11. Rather than using the public's estimate of benefits (in terms of willingness to pay) the regulator has to come to a view as to what cost is reasonable to save a life or reduce a health risk. Equivalent kinds of money-cost relative to health-benefit comparisons are also made in the US regulatory system.

The manner in which technology-based instruments have been implemented varies considerably between countries. In the UK, officials of the Inspectorate of Pollution negotiate controls with plant managers, but have the right, in the last instance, to require the adoption of certain control technologies. The United States Environmental Protection Agency administers a rather more uniform control programme: in 1990, Congress required the EPA to establish technology-based standards for about 200 specific pollutants.

instrument does not focus abatement effort on polluters that can abate at least cost. Moreover, there is an additional inefficiency here that also involves information asymmetries, and which relates back to a point made earlier about Figure 6.2. Technology requirements restrict the choice set allowed to firms to reduce emissions. Decisions about emissions reduction are effectively being centralised (to the EPA) when they may be better left to the firms (who will choose this method of reducing emissions rather than any other only if it is least-cost for them to do so).

Required technology controls blur the pollution target/pollution instrument distinction we have been drawing in this and the previous chapter. The target actually achieved tends to emerge jointly with the administration of the instrument. We need to be a little careful here. Sometimes government sets a general target (such as the reduction of particulates from diesel engines by 25% over the next 5 years) and then pursues that target using a variety of instruments applied at varying rates of intensity over time. In this case, no single instrument need necessarily have a particular target quantity associated with it. Nevertheless, it does matter (as far as cost-efficiency is concerned) if the actual operation of any particular component of this programme does not involve any comparison of the benefits and costs of that component (because then the wrong mix of components will be used). There are many examples of technology control where it appears to be the case that emphasis is given almost exclusively to the costs of pollution reduction technologies, and in particular to what kind of cost premium is involved in using the technically best method as compared with its lower-ranked alternatives, without any particular regard to the expected benefits that a superior technology will bring.

Although technology-based instruments may be lacking in cost-effectiveness terms, they can be very powerful; they are sometimes capable of achieving large reductions in emissions quickly, particularly when technological 'fixes' are available but not widely adopted. Technology controls have almost certainly resulted in huge reductions in pollution levels compared with what would be expected in their absence.

6.3.2.3 Location

Pollution control objectives, in so far as they are concerned only with reducing human exposure to pollutants, could be met by moving affected persons to areas away from pollution sources. This is only relevant where the pollutant is not uniformly mixing, so that its effects are spatially differentiated. Implementing this *ex ante*, by zoning or planning decision, is relatively common. *Ex post* relocation decisions are rarer because of their draconian nature. There have been examples of people being removed from heavily contaminated areas, including movements away from irradiated sites such as Chernobyl, Times Beach (Missouri) and Love Canal (New York). However, it has been far more common to move pollution sources away from areas where people will be affected, or to use planning regulations to ensure separation. Planning controls and other forms of direct regulation directed at location have a large role to play in the control of pollution with localised impacts and for mobile source pollution. They are also used to prevent harmful spatial clustering of emission sources.

Location decisions of this kind will not be appropriate in many circumstances. Moving people away from a pollution source cannot, for example, reduce impacts on ecosystems. Relocating (or planning the location of new) emission sources has wider applicability, but will be of no use in cases where pollution is uniformly mixing. In Section 6.5 we shall consider a number of incentive-based instruments that are designed, among other things, to influence the spatial location of emissions sources. These are not, however, examples of command and control instruments.

6.3.2.4 Command and control instruments: a review

Our discussions in Section 6.3 have broadly led to the conclusion that whilst command-and-control (CAC) instruments may have attractive properties in terms of certainty of outcome, and power to get desired results very quickly, they are likely to do so in an inefficient way. This inefficiency arises from the fact that CAC techniques contain no mechanisms to bring about two desired results:

1. in any single programme of emissions reduction, equalization of marginal abatement costs over the controlled firms in that programme;
2. between different programmes seeking to achieve equivalent objectives, equalization of marginal abatement costs in attaining that over the different programmes.

The first of these two desired results – equalization of marginal abatement costs over all controlled firms in a single pollutant control programme – is one we examined earlier in Section 6.2 when introducing the idea of cost-effectiveness. Data that might indicate the magnitude of efficiency losses from a failure to equalise marginal costs in this case are unfortunately both scarce and unreliable, as it requires information about firm-level marginal abatement costs. It is easier, though, to get some handle on the potential size of inefficiencies arising from failure to equalise marginal costs across differing programmes that seek the same objective. Magat *et al*. (1986) estimated the marginal treatment cost of biological oxygen demand (BOD) from US rivers and lakes. The authors estimated that marginal costs of attaining regulatory standards varied from as little as $0.10 per kilogram of BOD removal to as much as $3.15.

As a second example, we look at some studies that investigate the relative costs of achieving human mortality reductions by various CAC programmes operated by the United States Environmental Protection Agency (US EPA). Van Houtven and Cropper (1996) examined several areas over which the US EPA has sought to achieve mortality reductions by CAC regulation. One such area arises from US EPA bans on particular uses of asbestos. Van Houtven and Cropper investigated 39 applications for asbestos use. From data on the costs of regulation and the number of lives expected to be saved in each application, the authors were able to estimate the value of a statistical life that is implied by US EPA decisions. By definition, if an action results in the expected level of deaths falling by one person over some relevant time period, that action has saved one statistical life. Van Houtven and Cropper found that, on average, products were banned when the cost of saving one life was below $49 million (in 1989 US dollar prices).

Van Houtven and Cropper obtain a very similar implied value ($51.51) million for a fatal cancer avoided in their study of 245 pesticide applications (of which 96 were banned). Decisions here were taken under the auspices of FIFRA agricultural chemicals legislation, which requires cost and benefit balancing to be used by the US EPA. A third area studied by Van Houtven and Cropper concerned CAC controls of toxic air pollutants – specifically benzene, arsenic, asbestos and mercury – under the provisions of the Clean Air Act. Prior to 1987, the implied value of a fatal cancer avoided was about $16 million. In 1987, a Court of Appeals ruling found that EPA has been unlawfully considering the costs of regulation in making its decisions. In so doing, some emissions had been allowed where the US EPA had estimated the cost-to-benefit ratio to be too high to justify control. The tighter standards the US EPA subsequently imposed (based only on the benefits of control) implied a value of a statistical life after 1987 of $194 million.

These estimates show wide variation ($16m, $49m, $51m, $194m). But even the lowest estimate is considerably higher than the values which individuals seem to be willing to pay to reduce the risk of death. For example, Viscusi (1992, 1993) estimated the compensating wage differential required by workers to take on high-risk jobs. Observed wage differentials imply a value of a statistical life of $5 million, just one-tenth of that implied by US EPA regulations under FIFRA legislation.

As a final example, which again casts doubt on the equalisation of control costs per unit benefit achieved, we present results taken from Viscusi (1996), which examines a number of command and control regulations designed to save lives and protect health. Table 6.3 shows the costs of a statistical life saved for each category of regulation. Huge variability is evident, although some of this reflects differences in what the US EPA is required to consider in making decisions (that is: just benefits, benefits and costs but without balancing, or benefits and costs with balancing).

We must be a little guarded in the conclusions drawn from the results given in this section. Most of those results give average cost relativities rather than marginal cost relativities, the latter being the one we ideally need for purposes of establishing cost-inefficiency. Nevertheless, in the case of both BOD removal and reduction of the risk of death, the magnitudes of variation between figures from different programmes suggest that there might be very large efficiency gains possible from reallocating control (and so control expenditures) from high-cost to low-cost areas. If this inference is correct, one might expect to see government progressively moving away from the use of CAC instruments towards incentive-based instruments, because (as we shall see in the next section) the latter do have better cost-efficiency properties. Environmental control in the USA certainly seems to be moving that way, as shown in Box 6.3.

Table 6.3 The statistical value of a life as revealed by US EPA command and control regulations

Regulation	Initial annual risk	Expected annual lives saved	Cost per expected life saved ($US 1984)
Unvented space heaters	2.7 in 105	63.000	0.10
Airplane cabin fire protection	6.5 in 108	15.000	0.20
Auto passive restraints/belts	9.1 in 105	1850.000	0.30
Underground construction	1.6 in 103	8.100	0.30
Servicing wheel rims	1.4 in 105	2.300	0.50
Aircraft seat cushion flammability	1.6 in 107	37.000	0.60
Aircraft floor emergency lighting	2.2 in 108	5.000	0.70
Crane suspended personnel platform	1.8 in 103	5.000	1.20
Concrete and masonry construction	1.4 in 105	6.500	1.40
Benzene/fugitive emissions	2.1 in 105	0.310	2.80
Grain dust	2.1 in 104	4.000	5.30
Radionuclides/uranium mines	1.4 in 104	1.100	6.90
Benzene in workplace	8.8 in 104	3.800	17.10
Ethylene oxide in workplace	4.4 in 105	2.800	25.60
Arsenic/copper smelter	9.0 in 104	0.060	26.50
Uranium mill tailings, active	4.3 in 104	2.100	53.00
Asbestos in workplace	6.7 in 105	74.700	89.30
Arsenic/glass manufacturing	3.8 in 105	0.250	142.00
Radionuclides/DOE facilities	4.3 in 106	0.001	210.00
Benzene/ethylbenzenol styrene	2.0 in 106	0.006	483.00
Formaldehyde in workplace	6.8 in 107	0.010	72000.00

Source: Viscusi (1996), pp. 124–125

Box 6.3 Evolution in the USA from command and control towards incentive-based instruments for environmental regulation

A recent USA EPA report ('The United States Experience with Economic Incentives for Pollution Control'), available online at http://yosemite.epa.gov/ee/epa/eed.nsf/Webpages/USExperienceWithEconomicIncentives.html, noted two recent trends in the form of instruments used in the USA for environmental management. The first is an increasing diversity of economic incentives used by EPA. Historically EPA had relied on regulations to reduce pollution and improve the environment, but it has begun to use a wide variety of economic incentive mechanisms in recent years. Second, the number of applications of incentive-based instruments at other levels of government – at the state and local level – has grown rapidly.

The Report suggested several reasons for these trends:

- Incentives are proving to be particularly useful in controlling pollution that has not already been subjected to traditional forms of regulation. For example, volume-based disposal charges encourage households to reduce solid waste by recycling, composting and other means.
- Incentive-based instruments can sometimes be applied where traditional regulations might not be possible. They are particularly useful for small and geographically dispersed sources.
- Incentives generate benefits beyond what is possible with traditional regulations. Two benefits stand out. First, they provide cost savings relative to command and control (CAC). One study estimates potential savings of widespread use of economic incentives could reach $45 billion annually. On a practical level, acid rain trading savings are at least $700 million annually. Second, incentives can provide impetus for technological change.

The United States system of environmental controls is one of the most comprehensive to be found. A set of Congressional statutes provides the legal framework for the regulatory system, and give responsibility to the United States Environmental Protection Agency (US EPA) for implementing and administering the system. A comprehensive, and well-indexed, account of US environmental policy can be found on the 'Laws

Box 6.3 *continued*

& regulations' section of the US EPA website, http://www.epa.gov/lawsregs/. Here we focus on a number of important areas within that regulatory framework: air and water pollution, hazardous waste disposal, agricultural chemicals, toxic substances, and species protection. What is evident is that a framework initially based around conventional command and control regulations is increasingly characterised by the use of incentive-based instruments.

Air quality

The *Clean Air Act* defines ambient air quality standards for all parts of the USA for two types of pollutant: criteria (common) and hazardous air pollutants. *Criteria* air pollutants consist of particulates, SO_2, CO, NO_2, low-level ozone and lead. Each of these is given a *primary NAAQS* (National Ambient Air Quality Standard), set to protect human health. Some are also given a *secondary* NAAQS to protect wildlife, visibility and ecological systems. The levels of NAAQS for the criteria pollutants were listed in Table 5.4 in Chapter 5.

The system for *criteria* air pollutants is as follows. For *stationary sources* of air pollutants, the principal control instrument has been technology-based regulation, supported by maximum allowable emissions rates in some cases. *Existing pollution sources* must satisfy 'reasonably available control technology' (RACT). *New pollution sources* must meet more restrictive 'new source performance standards' (NSPS), based on the criterion of commercially available 'best technological system of emissions reduction'. Where NAAQS have not been met, stricter criteria may be used, such as 'lowest achievable emissions rate' (LAER), or in Class 1 (unspoilt) areas 'best available control technology' (BACT). Firms may be required to use particular techniques to recover fumes or waste products, or they may be prohibited from using certain production processes. Not surprisingly, the interpretation of these different criteria and the particular requirements that US EPA mandates for firms are contentious, and lead to significant amounts of judicial action.

For *mobile source* air pollution, control is largely directed at vehicle manufacturers, again in the form of required technology controls. Stricter controls are used in some non-attainment areas (such as mandated use of low-polluting fuels).

Although air pollution is mainly controlled by technology-based regulation, there are some exceptions. A flexible incentive-based system has been developed for acid-rain-inducing pollutants. Some states are experimenting with various market-based controls, such as those being used in the Los Angeles basin area.

In the cases of *hazardous air pollutants* (about 200 air toxins listed by US EPA), 'large' stationary sources must use 'maximum achievable control technology' (MACT). With the passage of time, US EPA has gone some way along the process of defining acceptable risk in operational terms.

The Acid Rain Program represents a dramatic departure from traditional command and control regulatory methods that establish specific, inflexible emissions limitations with which all affected sources must comply. Instead, the Acid Rain Program introduces an allowance-trading system that harnesses the incentives of the free market to reduce pollution:

- trading of sulfur dioxide allowances in the Acid Rain program, which encourages utilities to find least cost compliance strategies;
- basing air emission permit fees on the quantity of emissions.

Clean water

Water standards are again typically based on technology controls. In the initial control phase, this required the use of 'best *practical* technology' (BPT). Later control phases mandated the more stringent 'best *available* technology' (BAT). In addition to BAT, dischargers must acquire (non-marketable) effluent emissions licences, often containing very detailed plans about how discharges are treated as well as the amounts that may be discharged. Technology controls ('best-management practices') are also employed to reduce runoff from non-point sources (industrial and agricultural sites).

- Charging for the disposal of industrial effluents in water treatment plants.

Hazardous waste disposal

Under the terms of the Resource Conservation and Recovery Act, the US EPA has developed a list of about 450 hazardous substances. Disposal

Box 6.3 *continued*

is controlled through location restrictions, required staff training, groundwater monitoring by disposing firms, and the requirement to construct detailed plans for site closure and post-closure practice. Operators must also undertake sufficient insurance cover. These, and other, restrictions are supported by a licence system. The Superfund has provided a mechanism for dealing with abandoned waste dumps. The fund is built up from general taxation and from taxes on the petroleum and chemical industries. The principle of 'strict, joint and several liability' establishes strong incentives throughout the waste cycle to minimise the amount of waste produced.

Toxic substances

The TSCA requires US EPA to review all new chemicals, and gives it authority to restrict use of, or ban, existing chemicals. Unlike most areas of environmental regulation, the TSCA requires balancing of the costs of regulation (in money terms) and the benefits of regulation (in terms of cancer or other serious health impacts avoided).

- Encouraging reductions in toxic emissions by broadly disseminating information about emissions through hazard warning labeling and in communities through the annual Toxics Release Inventory.

Agricultural chemicals

FIFRA imposes a duty of registration of all new pesticides. New ingredients in agricultural chemicals cannot be introduced until the US EPA is satisfied, after cost–benefit analysis, that the product will generate positive net benefits. EPA may also carry out Special Reviews on existing pesticides. As with TSCA, FIFRA requires that the EPA 'balance' benefits against costs in arriving at its decisions about bans or other restrictions. A study by Van Houtven and Cropper (1996) investigated 245 food crop applications of 19 pesticide active ingredients. Of these, 96 applications were banned after US EPA Special Reviews.

Examples of economic incentives discussed in the report include:

- subsidising farmers and others to conserve habitat and control pollution;
- requiring a deposit on beverage containers to encourage recycling, which now occurs in ten states; many states have a similar system for lead acid batteries;
- imposing liability for natural resource damages caused by oil and hazardous material spills, a incentive to encourage pollution prevention;
- promoting voluntary programs such as Energy Star, Waste Wise, XL and other programs that reduce pollution by assisting and rewarding voluntary actions to reduce energy use and limit pollution.

6.4 Economic incentive (quasi-market) instruments

Command and control instruments operate by imposing *mandatory* obligations or restrictions on the behaviour of firms and individuals. In contrast, incentive-based instruments work by altering the structure of pay-offs that agents face, thereby creating incentives for individuals or firms to *voluntarily* change their behaviour. The pay-off structures are altered by changing relative prices. This can be done in many ways.[12] We focus on two of them in this section:

1. by the imposition of taxes on polluting emissions (or on outputs or activities deemed to be environmentally harmful), or by the payment of subsidies for emissions abatement (or reduction of outputs or activities deemed to be environmentally harmful);

[12] The forms of incentive-based instruments that we focus on in this section are taxes (or other charges) on emissions, emissions abatement subsidies, and marketable emissions permits. But it follows that any instrument that manipulates the price system in such a way as to alter relative prices could also be regarded as an incentive-based instrument. Other forms, some of which are considered in this chapter or elsewhere in the book, include deposit-refund systems, liability payments, non-compliance fees, charges on landfill or other disposal of waste, and performance bonds.

2. by the use of tradable emission permit (or allowance) systems. Companies, or other controlled parties, are entitled to emit designated pollutants up to the quantity of allowance that they hold. Those allowances – summing in total to whatever aggregate target the EPA seeks to achieve – are distributed without charge to potential pollution emitters or are sold by auction. Given that allowances are tradable, permit markets will emerge with associated market prices. Those prices are, in effect, the cost of emitting pollutants.

Whichever of these two ways is chosen, prices exist which generate opportunity costs that profit-maximising firms or utility maximising individuals will take account of and so the instrument generates incentives to make behaviour less polluting.[13] We now examine these two approaches, looking first at the use of tax and subsidy instruments.

6.4.1 Emissions taxes and pollution abatement subsidies

In this section, we examine a tax on emissions, or a subsidy on emissions abatement. For simplicity, we begin with the special case of uniformly mixed pollutants, for which the magnitude of damage created by an emission is independent of the location of its source. It is shown later that the results also apply, with minor amendment, to non-uniformly-mixing pollutants. Given that taxes on emissions are equivalent to subsidies (negative taxes) on emissions abatement, it will be convenient to deal explicitly with tax instruments, and refer to subsidy schemes only when there is a difference that matters.

Looking again at Figure 6.4, it is evident that there are several points at which a tax could be applied (just as there were several points of intervention for command and control regulations).[14] We focus here on emissions taxes. It is important to note that taxes on output of the final product will not have the same effect as emissions taxes, and will generally be less efficient in attaining pollution targets. This matter is examined in Appendix 6.1 and in Problem 9 at the end of the chapter.

A tax on pollutant emissions has for long been the instrument advocated by economists to achieve a pollution target. It is useful to distinguish between three cases:

1. the pollution target is the economically efficient level of pollution (the level which maximises social net benefits);
2. a specific target is sought, but it is set according to some criterion other than economic efficiency;
3. the EPA has insufficient information to know how much emissions reduction will be obtained from any particular tax rate, but seeks an emission reduction of some unknown amount by arbitrarily selecting some emission tax rate.

We deal with each of these cases in turn. To attain the *efficient level* of pollution, it is necessary to have solved the net benefit maximisation problem discussed in the previous chapter. A shadow price for emissions implicitly emerges from that exercise, this price being equal to the monetary value of marginal damage from emissions at the socially efficient level of pollution. This is the rate at which the tax (or subsidy) should be applied per unit of emissions.

Figure 6.5 illustrates the working of an emissions tax. Note that the diagram uses aggregate, economy-wide marginal benefit and marginal damage functions (not those of individual firms). In the absence of an emissions tax, if firms behave without regard to the pollution they generate, emissions will be produced to the point where the private marginal benefit of emissions is zero. This is shown as $\hat{M}$, the pre-tax level of emissions.

Now suppose an emissions tax was introduced at the constant rate μ^* per unit emission, the value of

[13] Liability can also be viewed as an incentive-based instrument, although we do not pursue that interpretation any further here.

[14] In fact, taxes could be levied even 'earlier' in the supply chain than indicated by the categories shown in Figure 6.4. Thinking about carbon dioxide emissions, for example, instead of levying taxes on CO_2 emissions at the production stage (where emissions are directly emitted) one could instead apply a tax on fossil fuel extracted at the resource extraction stage, where the tax rate is chosen to reflect the social cost of CO_2 emissions. A tax at this stage would cascade through the economy, changing the patterns of relative prices. This approach is examined in Chapter 8.

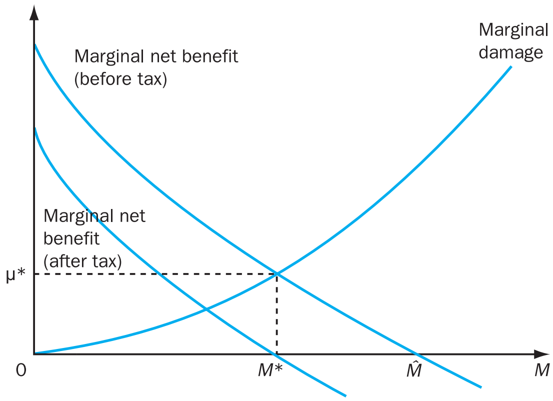


Figure 6.5 An economically efficient emissions tax

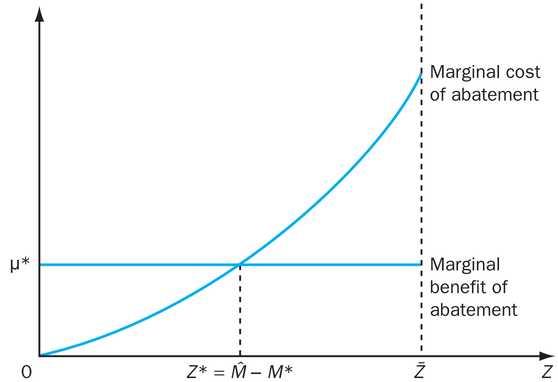


Figure 6.6 The economically efficient level of emissions abatement

marginal damage at the socially efficient pollution level. Given this, the post-tax marginal net benefit schedule differs from its pre-tax counterpart by that value of marginal damage. Once the tax is operative, profit-maximising behaviour by firms leads to a pollution choice of M^* (where the post-tax marginal net benefits of additional pollution are zero) rather than $\hat{M}$ as was the case before the tax. Note that levying an emissions tax at the rate μ^* creates just the right amount of incentive to bring about the targeted efficient emission level, M^*.[15]

It is sometimes more convenient to view the problem in terms of emissions abatement, Z, rather than the level of emissions, M. This can be done by reinterpreting Figure 6.5. Viewed in this new light, the emission tax causes abatement to increase from zero (at $\hat{M}$) to its efficient level $Z^* = \hat{M} - M^*$ at the point M^* on the horizontal axis of Figure 6.5. Alternatively, we can map the relevant parts of Figure 6.5 into abatement space, from which we obtain Figure 6.6.

It is important to be clear about the relationships between these two diagrams. First, the curve labelled 'marginal cost of abatement' in Figure 6.6 is just the mirror image of the (before-tax) marginal net benefit curve in Figure 6.5; what firms privately forgo when they abate emissions is, of course, identical to the benefits they receive from emissions. The 'marginal benefit of abatement' to a representative firm is the tax rate applied, μ^*. Each unit of abated emissions reduces the firm's total tax bill by that amount. As the tax rate is constant, the marginal benefit of abatement curve is horizontal. Secondly, note that we have truncated the two curves in Figure 6.6 at $Z = \bar{Z}$, where $\bar{Z}$ is identical in magnitude to $\hat{M}$. (You should confirm for yourself the reason for doing this.) Finally, note that $Z^* = \hat{M} - M^*$, and so the distance from the origin to Z^* in Figure 6.6 is equal to the horizontal distance between $\hat{M}$ and M^* along the emissions axis in Figure 6.5.

In the absence of an emissions tax (or an abatement subsidy), firms have no economic incentive to abate pollution. (In terms of Figure 6.6, the marginal benefit of abatement lies at zero along the Z axis.) Profit-maximising behaviour implies that firms would then undertake zero abatement, corresponding to emissions $\hat{M}$. However, when an emissions tax is levied (or, equivalently, when an abatement subsidy is available) an incentive to abate exists in the form of tax avoided (or subsidy gained). It will be profitable for firms to reduce pollution as long as their marginal abatement costs are less than the value of the tax rate per unit of pollution (or less than the subsidy per unit of emission abated). If the tax/subsidy is levied at the level μ^* the efficient pollution level is attained without coercion, purely

[15] As shown in Appendix 6.1, a subsidy at the rate μ^* on units of pollution abated would have an equal short-run effect on emissions to a pollution tax at the rate μ^* on unabated units of pollution.

as a result of the altered structure of incentives facing firms.

In the language of externalities theory, the tax eliminates the wedge (created by pollution damage) between private and socially efficient prices; the tax brings private prices of emissions (zero, before the introduction of the tax) into line with social prices (μ^*). The tax 'internalises the externality' by inducing the pollution generator to behave as if pollution costs entered its private cost functions. Decisions will then reflect all relevant costs, rather than just the producer's private costs, and so the profit-maximising pollution level will coincide with the socially efficient level.

Not only will the tax instrument (at rate μ^*) bring about a socially efficient *aggregate* level of pollution but it will also achieve that aggregate target in a cost-effective way. Remember that cost-efficiency requires that the marginal abatement cost be equal over all abaters. Under the tax regime all firms adjust their firm-specific abatement levels to equate their marginal abatement cost with the tax rate. But as the tax rate is identical for all firms, so are their marginal costs.

This is a remarkable result. Knowledge of both the aggregate marginal pollution damage function and the aggregate emissions abatement cost function are necessary for achieving a socially efficient emissions target at least real resource cost to the economy as a whole. But it is ***not*** necessary to know *each firm's* marginal abatement cost function. Compare this result with the case of command and control instruments where attaining an aggregate target at least real resource cost *does* need that additional, and far more demanding, information set: each firm's marginal abatement cost function.

Our discussion in this section so far has dealt with the case in which the EPA wishes to attain the economically efficient level of emissions, M^*. However, the EPA may not have sufficient information for this to be feasible, or it may wish to set an overall emissions target on some other basis. Suppose that the EPA does have an emissions target, $\tilde{M}$, set perhaps on health grounds. In terms of diagrams, we now need a slightly different version to that shown in Figure 6.5. The most likely reason that the EPA does not have sufficient information to use an economically efficient target is that it does not

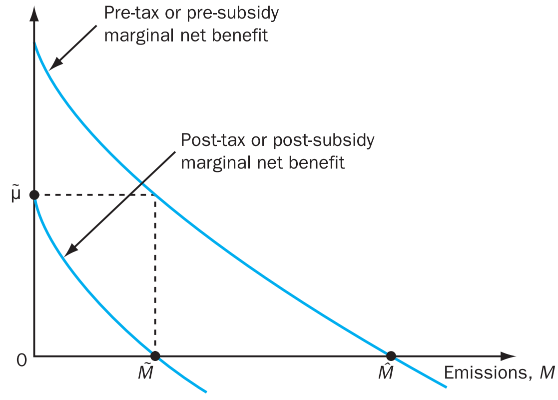


Figure 6.7 Emissions tax and abatement subsidy schemes when marginal damage is unknown, or when a target is being set on grounds other than economic efficiency

know the location of the marginal damage function. Alternatively, if the EPA sets a target on some ground other than economic efficiency, the marginal damage function as shown in Figure 6.5 is being treated as irrelevant, and a target $\tilde{M}$ is being set exogenously. Figure 6.7 illustrates. Figure 6.7 has been drawn such that the exogenously set emissions target, $\tilde{M}$, is more strict than the efficient target in 6.5.

Figure 6.7 makes it clear that to attain this (or indeed any other specific) emissions target using a tax or subsidy instrument, knowledge of the aggregate (pre-tax or pre-subsidy) marginal benefit of emissions function would be sufficient. For any target $\tilde{M}$, the location of that function allows the EPA to identify the tax rate, say $\tilde{\mu}$, that would create the right incentive to bring about $\tilde{M}$. Note that as Figure 6.7 is drawn to show a target more strict than M^*, the shadow price of emissions (the implied tax rate or unit emission subsidy) $\tilde{\mu}$ is higher than μ^*.

By construction, the marginal net benefit of emissions function is exactly equivalent to the marginal abatement cost function. So the reasoning given in the previous paragraph could be reworded so that it applies to Figure 6.6. Thus, for any target $\tilde{M}$, knowledge of the aggregate marginal cost of abatement function allows the EPA to identify the tax rate, say $\tilde{\mu}$, that would create the right incentive to bring about that outcome.

Note that even though the target is not a socially efficient target, the argument used earlier about cost-efficiency remains true here: the emissions tax, levied at $\tilde{\mu}$, attains the target $\tilde{M}$ at least total cost, and so is cost-efficient. This result is rather powerful. To achieve any arbitrary aggregate target at least cost the EPA does *not* need to know the aggregate marginal pollution damage function; knowledge of the aggregate abatement cost function *alone* is sufficient for achieving the target at least cost. And again, as in the case of an economically efficient emissions target, the EPA does ***not*** need to know the abatement cost function of each firm. The least-cost condition of equalised marginal abatement cost is 'automatically' ensured by the fact that a uniform tax rate is being applied over all emitters.

Finally, let us deal with the third of the listed cases where an emission reduction of some unspecified amount is sought. If the EPA knew the location of neither the aggregate marginal pollution damage function nor the aggregate emissions abatement cost function, it would have insufficient information to know how much emissions reduction would be obtained from the imposition of any specific tax rate. But the EPA might still select some arbitrary positive level of emissions tax, say £8 per tonne of the controlled pollutant, knowing that some degree of reduction will be achieved.[16] Faced with this tax rate, profit-maximising firms will reduce emissions up to the point where marginal abatement costs are brought into equality with this tax rate. As all firms do this, the emissions reduction is achieved at least real resource cost once again.[17] Although the government cannot know in advance how much pollution reduction will take place, it can be confident that whatever level of abatement is generated would be attained at minimum feasible cost. Taxes (and subsidies by an equivalent argument) are, therefore, cost-efficient policy instruments. These results are demonstrated formally in Appendix 6.1, Parts 4 and 5.

6.4.1.1 Are pollution taxes and emissions abatement subsidies equivalent?

We stated earlier that an emissions tax and an emissions abatement subsidy (at the same rate) have an identical effect in terms of pollution outcome in the short term (see Part 6 of Appendix 6.1). However, the two instruments do have some very important differences. Most importantly, the distribution of gains and losses will differ. Taxes involve net transfers of income from polluters to government, while subsidies lead to net transfers in the other direction (see Problem 4). This has important implications for the political acceptability and the political feasibility of the instruments. It also could affect the long-run level of pollution abatement under some circumstances. Let us examine this matter more closely.

For an industry *of a given size*, an emission tax and an abatement subsidy levied or paid at the same rate are equivalent in terms of units of emissions abated. Thus, looking at Figure 6.5 again, a subsidy or a tax at the rate μ^* would reduce emissions from $\hat{M}$ to M^* for a single firm with a given capital structure. As the industry is simply the sum of all firms, if the number of firms remains constant and the capital structure of each firm is unchanged, then the effects of taxes and subsidies are identical.

However, the two instruments are different in their effects on income distribution. A firm gains additional income from an abatement subsidy, as it will undertake abatement only when the unit abatement subsidy exceeds its marginal abatement cost. A tax, on the other hand, results in a loss of income to the firm as it pays the tax on ***all*** its emissions. To make this comparison more precise, look at Figure 6.8, the functions in which reproduce those in Figure 6.5.

An abatement subsidy will result in a payment to the firm equal to the areas $S_1 + S_2$, that is, μ^* the subsidy rate multiplied by $(\hat{M} - M^*)$, the amount of emissions reduction. However, by reducing emissions

[16] We are, of course, comparing the case of an emissions tax being introduced with a 'baseline' case in which there is no emissions tax. Presumably, similar arguments would follow if an existing emissions tax were increased to a higher level.

[17] All references to the phrase 'least cost' in this chapter are abbreviated versions of the more precise term 'least real resource cost'. The reason for the qualifier 'real resource' is to make it clear that it is not financial costs that are being minimised but ***resource*** costs in real terms (real in the sense of being at real prices rather than market prices which may include taxes, subsidies or other transfer payments or receipts). By resource costs, we are referring to such things as the expenditures on pollution abatement equipment, the additional costs of using less polluting inputs, the opportunity costs of lost output, and the like.

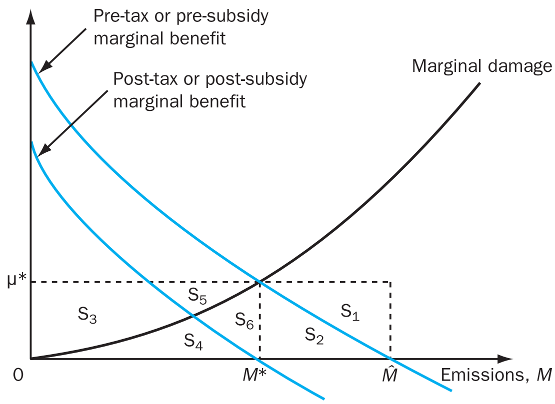


Figure 6.8 Emissions tax and abatement subsidy schemes: a comparison

from $\hat{M}$ to M^* the firm loses S_2 in profit on final output. The net *gain* to the firm is equal, therefore, to the area S_1.

In contrast, a tax levied at the rate μ^* on emissions M^* will cost the firm μ^*M^*, that is, the sum of the areas S_3, S_4, S_5 and S_6. However, by reducing emissions from $\hat{M}$ to M^* the firm also loses profit on reduced output, the area S_2. The net *loss* to the firm is given by the sum $S_3 + S_4 + S_5 + S_6 - S_2$. So the income effects are entirely different.

Let us explore this difference a little further. Recall that the tax paid is equal in value to μ^*M^*, while the subsidy received is $\mu^*(\hat{M} - M^*)$. But $\mu^*(\hat{M} - M^*) = \mu^*\hat{M} - \mu^*M^*$. The second term on the right-hand side is the tax paid, and will depend on the amount of abatement undertaken. It is this second component which gives the firm an incentive to abate emissions. Recalling that μ is an outflow in a tax scheme and an inflow in a subsidy scheme, an outflow of μ^*M^* (with a tax) is identical to an inflow of $-\mu^*M^*$ (with a subsidy). The two incentive effects are identical, and it is this that forms the basis for the claim that the instruments are equivalent. However, the subsidy differs from the tax by the presence of the additional term, $\mu^*\hat{M}$, a fixed or lump-sum payment, independent of the amount of abatement the firm actually undertakes. In the long run such payments may alter industry profitability, and so alter the size of the industry itself. This lump-sum payment component of the subsidy may destroy the equivalence between the two instruments in terms of their effects on emissions abatement.

We are faced with the possibility that a subsidy might enlarge the industry, partially or wholly offsetting the short-run emissions reduction. It is not possible to be more precise about the final outcome, as that depends on several other factors, including whether or not government introduces other fiscal changes to counteract the income effects we have just described. A general equilibrium analysis would be necessary to obtain clear results. This is beyond our scope in this text, so we just note that the equivalence asserted above is not valid in all cases.

Finally, note another aspect of an abatement subsidy scheme. As one component of the subsidy payment depends on the uncontrolled level of emissions (that is, the component $\mu^*\hat{M}$), a firm has an incentive to misrepresent the uncontrolled level of emissions in order to obtain a favourable benchmark in terms of which the subsidy payments are calculated.

To reinforce your understanding of this material in this section, you are recommended to work through Problem 10 at the end of this chapter. This uses an Excel workbook to simulate emissions reduction using command and control techniques, tax and subsidy instruments, and (to be discussed in the next section) transferable emissions permits. Some information on practical experience with pollution taxes and abatement subsidies is given in Box 6.4.

6.4.2 Marketable emissions permits

As with command and control and tax/subsidy instruments, marketable permits (also known as tradable or transferable permits) can be applied at many points in the production-to-pollution process. Here we consider only one form: permits on the quantity of directly produced *emissions*. Marketable permit systems are based on the principle than any increase in emissions must be offset by an equivalent decrease elsewhere. There is a limit set on the total quantity of emissions allowed, but the regulator does not attempt to determine how that total allowed quantity is allocated among individual sources.

There are two broad types of marketable emission permit system. The first type is a 'cap-and-trade' system. The second type we call a 'flexible permit

Box 6.4 Emissions taxes in practice

Fees, charges and taxes (terms we shall use interchangeably in this box) are the most commonly used economic instrument for environmental management. Fees affect environmental quality in two ways:

- By directly affecting polluting behaviour through the choice of inputs to firms and the production techniques they select, and through the choice of products purchased by households and how those products are used.
- By providing a source of revenue to pay for governmental environmental management or to subsidise pollution control activities.

Tax rates are typically set at levels insufficient to fully internalise external costs (EEA, 2000). Low rates of tax imply correspondingly low levels of impact. Typically fees are used mainly for revenue-raising purposes, or to cover the administrative and implementation costs of the programmes with which they are associated. One example is the use of taxes on new paint purchases in British Columbia to support reprocessing and safe disposal of used paint.

There are some cases, however, in which the main purpose of the fee is to control polluting emissions; it is these we focus on in this box. A survey by the United States Environmental Protection Agency (EPA, 2004) found evidence of a trend for taxes to be used in more applications and to be levied at higher rates. It also concluded that social and political acceptance of environmentally relevant charges was increasing.

Fees are often used in conjunction with standards and command and control instruments, rather than instead of those more conventional methods. In some cases, for example pollution taxes, are only levied on firms that fail to meet standards imposed on them.

Fee paying mechanisms are widely used for wastewater discharges and industrial effluent. In many cases, though, fees are set primarily to finance effluent treatment rather than to influence the magnitude of discharge. But many countries, including Brazil, China, France, Germany, Malaysia, the Netherlands, and the Philippines, have set fees high enough or related them sufficiently to potential damage to have a positive impact on environmental quality. In Germany, for example, significant increases in inflation-adjusted wastewater fees over the past 20 years have led to considerable technical innovation in water-saving dishwashers, washing machines and toilets. A user fee programme in Laguna de Bay, Malaysia, aimed at reducing the biological oxygen demand (BOD) of wastewater flowing into the lake, achieved a reduction in BOD from controlled sources of about 95% between 1997 and 2003. In 1998, France integrated several existing charges into a unified 'General Tax on Polluting Activities' (TGAP); the environmental agency is allocated a share of TGAP revenues for environmental improvement programmes, with a proportion of those revenues being repaid in the form of capital subsidies to firms adopting recommended control technologies.

The use of marketable permits to control air pollution has attracted much attention in the past decade. But far more common has been the use of taxes to control various air-polluting emissions. China taxes NO_x, SO_2, Cl_2, CS_2, CO, HCl, and fluoride emissions, and coal dust and cement dust in excess of standards. Tax rates vary between regions, being highest in areas with greatest air pollution concentrations. SO_2 emission charges are the highest in Beijing (in 2003 at 1.2 yuan/kg, equivalent to $150 per metric ton) and approximately the same price as SO_2 allowances in the US Acid Rain trading programme. If Chinese firms were all profit-oriented, the impacts of such fee levels would be very large. It is unclear, though, what kind of deterrent effects they have on state-owned not-for-profit organisations. Air pollutant emission fees thought to have substantial emission-reduction impacts are also employed in Japan (initially to generate revenues to compensate victims of pollution-related diseases) and France. Sweden charges combustion plants for NO_X emissions, with revenue being distributed among emitters in proportion to their share in total energy output. Hence the total cost of the system to emitters is zero, but each plant has an incentive to reduce its emissions-to-energy-output ratio. The regime appears to have led to significant falls in NO_X emissions and to have spurred innovation in combustion technology.

Several countries have imposed product charges on pesticides and fertilisers. Often cited cases include Sweden, Norway, Finland, Austria, and Denmark. In the last-mentioned case, Hansen (2001) estimated the price elasticity of demand for pesticides in Denmark at –0.45, which suggests that the 20% tax reduced pesticide use by 9%.

Box 6.4 *continued*

Norway imposed a product charge on purchases of trichloroethylene (TCE) and perchloroethelyne (PER). Sterner (2004), using preliminary data, showed that consumption of TCE fell from more than 500 tons in 1999 to 139 tons in 2001, while PER use fell from 270 tons in 1999 to 32 tons in 2001.

Energy or carbon-related taxes have been common in Europe, but the use of tax-based controls is increasingly being replaced by the use of marketable permits (see Box 6.5). Partly as a result of this switch of instrument choice, the European Union has abandoned plans for a common carbon tax. Taxes on fuels based on their carbon content have been adopted in Belgium, Denmark, Finland, France, Italy, Luxembourg, the Netherlands, Norway and Sweden. However, in the great majority of countries where CO_2 (or other environmental) taxes have been implemented, some sectors have been exempted from the tax, or the tax rate is differentiated across sectors. This reduces the cost-effectiveness, and so raises the real cost, of the tax. OECD (2004) provides a useful review of these and other energy taxes in OECD countries.

Many developed and developing countries have imposed higher taxes on leaded gasoline to encourage motorists to switch to unleaded fuel, with Austria, Denmark, Bulgaria, Denmark, Finland, France, Germany, Greece, Netherlands, Philippines, Poland, Portugal, Sweden, and the United Kingdom providing examples. (Also see Box 6.5 for an example of lead trading). Sweden and the UK have used differential charges and subsidies on cars and heavy vehicles to encourage the purchase of low-pollution engines and the adoption of catalytic converters. Municipal and other waste charges such as landfill taxes, and measures to encourage recycling, are examined in Box 6.7.

The USA makes little use of emissions taxes or charges, preferring recently to use marketable permits wherever that is practical (see Box 6.5). Exceptions include a tax on chlorofluorocarbons to help in the phasing out of these chemicals, and fees on sewage and solid and hazardous waste at landfills. Households typically pay by the gallon for sewage disposal, and waste haulage firms pay by the ton for solid waste disposal. However, household and business enterprises have traditionally paid *lump-sum* charges for solid waste disposal, and so *marginal* disposal costs are not passed on to the initial producers of waste, leading to significant efficiency losses. As more states move to volume-related charges (37 states now do this), volumes discarded have fallen and recycling rates have risen significantly (Anderson *et al.*, 1997).

Sources: Tietenberg (1990); Goodstein (1995); OECD (1999); International Experiences with Economic Incentives for Protecting the Environment. EPA (November 2004), at http://yosemite.epa.gov/ee/epa/eerm.nsf/vwRepNumLookup/EE-0487?OpenDocument, from which the contents of this box have drawn very heavily.

systems with offsets'.[18] Cap-and-trade permit systems are those discussed in the conventional textbook models of marketable permits, and can be thought of a 'pure' type. They are the form in which marketable permit systems were first discussed in the literature, were first introduced in practice, and are the more commonly found. We shall begin our exposition of marketable permits by analysing a generic cap and trade system in some depth. Flexible systems incorporating an offset principle will be considered more briefly in section 6.4.2.4 below.

A cap-and-trade marketable emission permits scheme *for a uniformly mixing pollutant* involves:[19]

- A decision as to the total quantity of emissions of some particular type that is to be allowed by a specified class of actual and potential emitters over some period of time. This total quantity is the 'cap'.
- The creation of a quantity of emissions permits that in sum equal, in units of permitted emissions, the emissions cap (the target level of emissions).
- A mechanism, chosen by the control authority, which determines how the total quantity of emission permits is initially allocated between potential polluters.

[18] This term is the authors' own expression as there seems to be no commonly agreed term in the literature for this type.

[19] We deal with marketable permits for non-uniformly-mixing pollutants in Section 6.5.3.

- A rule which states that no firm is allowed to emit pollution (of the designated type) beyond the quantity of emission permits it possesses.
- A system whereby actual emissions are monitored and penalties – of sufficient deterrent power – are applied to sources which emit in excess of the quantity of permits they hold.
- A guarantee that emission permits can be freely traded between firms at whichever price is agreed for that trade.

Marketable permit schemes differ from tax or subsidy schemes by working in terms of quantities rather than prices. But this feature is also true for command and control instruments such as quotas, licences and standards. The distinguishing feature is the transferability of permits between individual sources in the marketable permits case. Permit trading is not allowed in command and control quota or licence systems.

It is the exchange process that generates the attractive qualities of the marketable permit system. In effect, tradability creates a market in the right to pollute. In that market, the right to pollute will have a value, given by the prevailing market price. So the decision to pollute generates an opportunity cost. By emitting an extra unit of the pollutant, one unit of permit is used up and so cannot be sold to another firm. The firm incurs a cost in emitting each unit of the pollutant, that cost being the current market permit price. Intuitively, this suggests that a marketable permit system should be equivalent (at least in some ways) to a tax or subsidy system, provided the permit price is equal to the tax or subsidy rate. As we shall see, this intuition is correct.

6.4.2.1 The initial allocation of permits and the determination of the equilibrium market price of permits

The implementation of a marketable permits system requires that the EPA select a method by which the total allowable quantity of permits (the cap) is initially allocated among emission sources. Simplifying matters somewhat, it must choose one of the following:

- Case 1: the EPA sells all permits by auction.
- Case 2: the EPA allocates all permits at no charge (which in turn requires that a distribution rule be chosen).

The process by which an equilibrium permit price emerges differs in some respects between the two cases, although the equilibrium price itself will be identical. Let us consider first the case in which the EPA sells all permits by auction.

Case 1: Auctioned permits

Suppose that the permits are initially allocated through a competitive auction market. Individual firms submit bids to the EPA. When the received bids are ranked in descending order of bid price, the resulting schedule can be interpreted as a market demand curve for permits. Assuming that no strategic behaviour takes place in the bidding process, this market demand curve will be identical to the aggregate marginal benefits of emissions. However, as we showed earlier, the *marginal benefit of emissions* (MB_M) function is also interpretable as the mirror image of the *marginal cost of abatement* (MC_Z) function.[20] Hence, we can also say that the market demand curve will be identical to (a suitably redrawn) aggregate marginal cost of emissions abatement cost curve.

Why should that be the case? Think about matters from the point of view of one firm. Prior to entering the auction, a firm will own no emissions permits, and so will not be allowed to emit any positive value of the controlled pollutant. Clearly, this has potentially devastating profit implications for the firm, and might even entail it going out of business. If a firm were able to produce its intended good or service at any quantity it wished without producing any emissions, and it could do so without incurring any abatement cost (that is, pollution abatement is a free good for that firm), then it need not buy any permits. But that is a most unlikely scenario. In general (and as we have assumed in this chapter), organisations *will* incur abatements costs in reducing emission flows, and those costs will rise in total as the level of emissions reduction increases. Moreover, it is also

[20] See Figures 6.5 and 6.6, and the surrounding discussion.

likely that the *marginal* cost of emissions reduction (abatement) will rise as the quantity of abatement rises.

Thus the firm faces a trade-off. Starting from a trivially small output level of its intended good or service, producing additional output will bring in additional net income (revenue after subtracting production costs). But it will also generate additional costs; those costs will consist of *either* additional costs incurred in production by using cleaner fuels or superior technologies that are less polluting, *or* costs of buying the required emissions permits.

How does the firm choose which of these two alternatives to use? Cost-minimising behaviour by firms implies that permits will be purchased whenever the price to be paid for those permits is less than the additional abatement cost it would have to incur. To purchase a permit, of course, requires that a bid be placed to the permit auction, and that the bid be successful.

It is important to realise that we are reasoning here in terms of marginal quantities. Thus if a firm has an upward-sloping marginal abatement cost curve as in Figure 6.4 (the marginal cost of abatement increases with the quantity of total abatement), it will purchase an additional emissions permit whenever the marginal cost of abating emissions exceeds the permit price.

Putting all this information together, we can now reach the following conclusion. Rational firms will (typically) submit multiple bids to the auction. The bids it submits will have different offer prices. It will offer high prices to acquire some permits, as its marginal abatement cost is high. For other units of

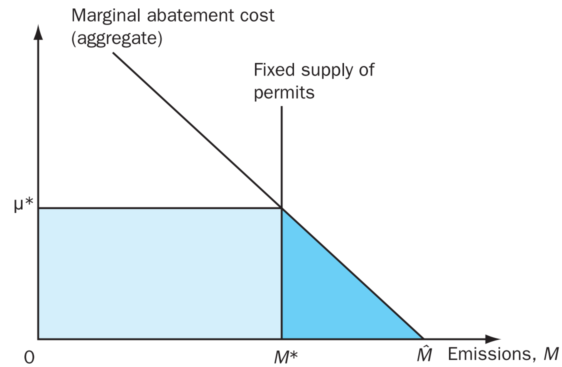


Figure 6.9 The determination of the market price of emissions permits

the permits, it will offer lower prices as its marginal abatement costs are lower. Each individual firm will, thus, have its own firm-specific demand curve for permits. Summing those individual demand curves over all firms who enter at least one positive bid will generate the *market* demand curve for permits in the auction market.

The market equilibrium permit price is determined by the value of the aggregate marginal abatement cost at the level of abatement implied by the total number of issued permits. This is illustrated in Figure 6.9. The market demand curve for permits is the aggregate marginal abatement cost function for all polluting firms. The total number of permits (allowed emissions) is M^*, and so the market supply of permits is shown in the diagram by a vertical line located above the quantity M^* on the horizontal axis. Given this quantity of permits, the market equilibrium price for permits will be μ^*.[21]

[21] This statement, and Figure 6.9, assume that all permits are sold at one price (the highest single price consistent with selling all permits, or equivalently the lowest price at which a successful bid was made). However, auction systems need not do this in practice. Another way of organising matters is for each successful bid to pay its actual bid price, so that prices paid by vary between μ* and the price offered in the highest bid (where the marginal abatement cost function reaches its highest point or meets the vertical axis). This does not affect the quantitative outcome of the market. The difference is simply distributional, as the latter method extracts the firms' producer surplus, and transfers it to the permits seller (the EPA). To give an example of this latter practice, SO_2 allowances are sold at successful bid prices in the USA. In the 2008 SO_2 Allowances Spot Auction, the following was observed:

Allowances	Number of Bids	Number of Bidders	Bid Price
Bid For: 599 370	Successful: 52	Successful: 17	Highest: $651.00
Sold: 125 000	Unsuccessful: 44	Unsuccessful: 2	**Clearing: $380.01** (the clearing price is the lowest price at which a successful bid was made)
	Total: 96	Total: 19	Lowest: $.27
			Weighted Average of Winning Bids: $389.91

Source: http://www.epa.gov/airmarkets/trading/2008/08summary.html (accessed 21 March 2009)

Case 2: Permits are allocated without charge to actual or potential polluters, using some kind of distribution rule

Next, we suppose that the EPA distributes the permits at no charge, and allow them to be subsequently traded in a free market. The initial allocation is unlikely to correspond to the desired (that is, profit-maximising) holdings of permits (and in aggregate, of course, should be less than total desired emissions if the instrument is to be binding). As you will see, although the distributional results differ considerably from Case 1, the analytics remain essentially the same, and the properties of the resulting resource allocation are identical.

Once this initial allocation has taken place, firms – both those holding permits in sufficient number to cover their desired emission levels and those not holding sufficient for that purpose – will evaluate the marginal worth of permits to themselves. These valuations will differ over firms.

Some firms hold more permits than the quantity of their desired emissions (in the absence of any control). The value of a marginal permit to these firms is zero.[22] Others hold permits in quantities insufficient for the emissions that they would have chosen in the absence of the permit system. The marginal valuations of permits to these firms will depend upon their emission abatement costs. Some will have high marginal abatement costs, and so are willing to pay high prices to purchase emissions permits. Others can abate cheaply, so that they are willing to pay only small sums to purchase permits; their marginal permit valuation is low.

In any situation where many units of a homogeneous product are held by individuals with substantially differing marginal valuations, a market for that product will emerge. In this case, the product is tradable permits, and the valuations differ because of marginal abatement cost differences between firms. Some firms, therefore, will try to buy additional permits from others, while others will try to sell some of their initial holding. Buyers will typically be firms with relatively high marginal abatement costs, who hope to purchase additional quantities at a price less than their marginal abatement cost. Sellers will be those in an opposite position, hoping to sell some permits at a price greater than their marginal abatement cost.

It is not necessarily the case that a firm which holds fewer permits than its desired emissions level will *buy* permits. It always has the option available to reduce its emissions to its permitted level by undertaking extra abatement. The firm may find it preferable to sell permits (rather than buy them) if the price at which they could be sold exceeds its marginal abatement cost.

Therefore, a market will become established for permits, and a single, equilibrium market price will emerge, say μ*. In a well-functioning competitive market, the market price that would emerge in this case would be identical to that which would be established if permits were sold at a competitive auction. This is portrayed in Figure 6.10. Note that the quantity traded, EP*, is less than the number of permits issued by the EPA (M^*), because trades only take place as holdings are adjusted to desired levels. Notice also that trading does not alter the quantity of permits in existence, it merely redistributes that fixed amount between firms. What is important to

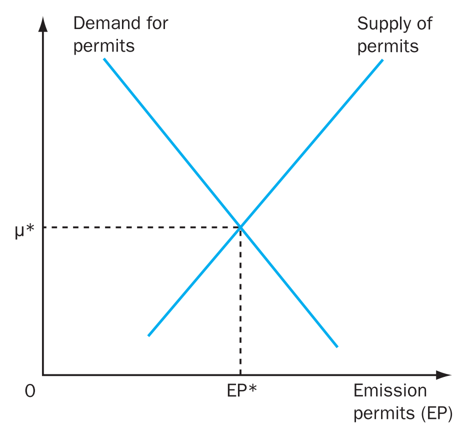


Figure 6.10 The determination of the market price of emissions permits: free initial allocation case

[22] If permits were storable or 'bankable' so that they could be used in the future, their worth would be positive (rather than zero) as there will be some positive probability that they could be used later when the firm would otherwise have insufficient permits to cover desired emissions. But we shall leave this complication to one side for now.

recognise from this analysis is that the method by which permits are initially allocated has no bearing on the amount of abatement that takes place; that depends only on the total number of permits issued. So for example, equilibrium permit prices and total quantity of emissions reduction will be identical for any initial basis of allocation, whether that be 'grandfathering' (where allocations are proportional to historical emissions levels), or equal allocations for all, or any other distributive principle.

In equilibrium marginal abatement costs will be equal over all firms, as they will with the case of auctioned permits too. It is this property of the system which ensures that transferable marketable permits, like taxes and subsidies, achieve any given target at least cost. Moreover, another equivalence arises. If the total quantity of permits issued is M^* *and* that quantity is identical to the level of emissions which would emerge from an emissions tax (or an abatement subsidy) at the rate μ^* then a marketable permit scheme will generate an equilibrium permit price μ^*. In effect, the marketable permit system is an equivalent instrument to either emissions taxes or emissions abatement subsidies. We demonstrate this result algebraically in Part 7 of Appendix 6.1.

There is one important qualification to these remarks about permit price determination. We have assumed that the permits market behaves as if it were perfectly competitive. But if the polluting industry in question is dominated by a small number of firms, or if for any reason the quantity of trading is small, strategic behaviour may take place. This could happen both in permit auctions and where firms are adjusting permit holdings from their initial allocations to their profit-maximising levels. Strategic behaviour may cause the market price of permits to diverge from its competitive level.

A numerical illustration

A simple numerical illustration (which extends an example used earlier in the chapter) will help to strengthen understanding about the way that marketable permits operate. Consider the information shown in Table 6.4. We suppose that the EPA selects an emissions cap – and so a total permit allocation – of 50 units. The pollutant is emitted by just two firms, A and B, and emissions abatement can only be undertaken by these firms. The EPA decides arbitrarily to allocate half of total permits to each firm, so prior to trading A and B are each allowed to emit 25 units of the pollutant. As in our earlier discussion, we assume that in the absence of any control system A would choose to emit 40 units and B 50 units. Given the initial permit allocations, A must reduce emissions by 15 units and B by 25 units. It can be seen from Figure 6.11 (which reproduces exactly the abatement cost functions used previously in Figure 6.1) that A has a marginal abatement cost of 45 and B a marginal abatement cost of 125.

Table 6.4 Emissions abatement data for firms A and B

	A	B	A + B
Uncontrolled emissions	40	50	90
Uncontrolled abatement	0	0	0
Efficient emissions	15	35	50
Efficient abatement	25	15	40
Initial permit allocation	25	25	50
Final permit allocation	15	35	50

The fact that firm A has lower marginal abatement cost than firm B after the initial permit allocation implies that the total abatement of 40 units of emission is not being achieved at least cost. Moreover, B places a much higher value on an incremental permit than does A (125 as compared with 40). Thus the two will find it mutually beneficial to trade with one another in permits. What will be the outcome of this trade? If the market behaved as if it were a competitive market, an equilibrium market price of 75 would emerge. At that price, firm B (the high-cost abater) would buy permits and A (the low-cost abater) would sell permits. In fact, B would buy 10 permits from A at 75 each, because for each of those 10 permits, it would be paying less than it would cost the firm to abate the emissions instead. Conversely, A would sell 10 permits to B at 75 each, because for each of those 10 permits, it would be receiving more than it would cost the firm to abate the emissions instead.

Trading finishes at the point where A has 15 permits (10 less than its initial allocation) and B has 35 (10 more than its initial allocation). Marginal control costs are equalised across polluters, and the total cost of abating emissions by 40 units has thereby

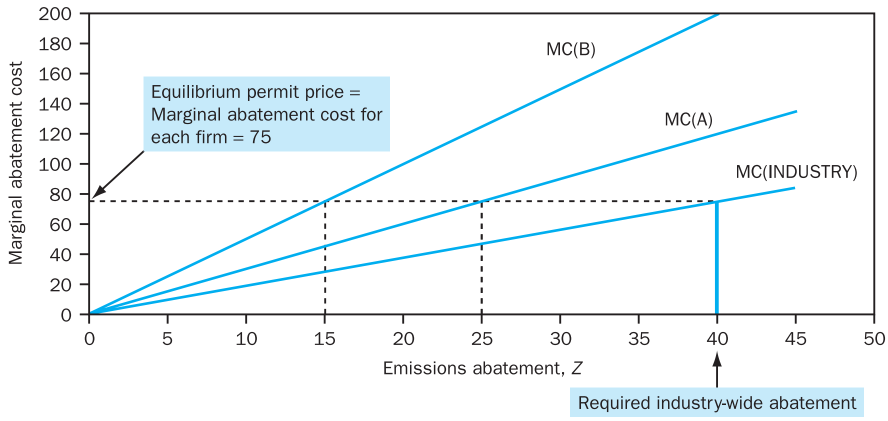


Figure 6.11 Efficient abatement with two firms and marketable permits

been minimised. The permit system will, therefore, have identical effects on output and emissions as an optimal tax or subsidy system, and will be identical in terms of its cost-effectiveness property.

One other feature shown in Figure 6.11 should be noted. The line labelled MC(Industry) is the industry-wide (or aggregate) marginal cost of abatement schedule. It is obtained by summing horizontally the two firm's marginal abatement cost functions, and is given by[23]

$$\text{MC(Industry)} = \frac{15}{8}Z$$

The equilibrium permit price is found as the industry marginal cost (75) at the required level of industry abatement (40). Note that as the required abatement rises, so will the equilibrium permit price.

6.4.2.2 Marketable permit systems and the distribution of income and wealth

In a perfectly functioning marketable permit system the method of initial allocation of permits has no effect on the short-run distribution of *emissions* between firms. But it does have significant effects on the distribution of *income and wealth* between firms. If the permits are sold by competitive auction, each permit purchased will involve a payment by the acquiring firm to the EPA equal to the equilibrium permit price. A sum of money equal to μ^* multiplied by M^* will thus be transferred from businesses to government.[24] This is shown by the lighter shaded area in Figure 6.9.

In addition to this, the emissions restrictions will impose a real resource cost (rather than a financial transfer) on firms. In terms of Figure 6.9 again, firms collectively are required to reduce emissions from $\hat{M}$ to M^* and so the real resource costs of the abatement are given by the area of the shaded triangle to the right of M^*; that is, the sum of marginal abatement costs over the interval $\hat{M}$ to M^*.

Note that the transfer of income from the business sector to the government when successful bids are paid for is not a real resource cost to the economy. No resources are being used, there is simply a transfer of income between sectors of the economy. Whenever we discuss least-cost methods of abatement in this chapter, you should note that it is the real resource costs that are being minimised, not any transfer costs such as those just referred to.

If, on the other hand, the EPA distributes permits at no charge, there is no transfer of income from

[23] To obtain this, first invert the two firm's functions, giving Z_A = (1/3)MC and Z_B = (1/5)MC. Next sum the two inverted equations to give Z = ((1/3) + (1/5))MC = (8/15)MC. Finally, invert this summed expression to obtain MC = (15/8)Z.

[24] An earlier footnote pointed out that if an auction system is operated so that each successful bid pays the actual bid price, then the transfer to government is larger, consisting not only of the lighter shaded area in Figure 6.9 but also the triangle that lies above that.

businesses to government. However, there will be transfers between firms. Some will buy from others and some will sell to others. So some firms will gain financially while others lose. The pattern and magnitude of these within-industry transfers will depend on the formula used to make the initial permit allocation. But even here there is still a real resource cost to the business sector, equal once again to the triangular shaded area in Figure 6.9. That burden is the same whatever initial allocation system is used. Taking all these remarks together, it is clear that the free allocation system is more attractive to polluting firms than the auction sale of permits.

The fact that there are different net income effects means that we must introduce the same qualification made earlier (in comparing taxes with subsidies) about long-run effects. An industry may contract in the long run if permits must be initially purchased; this effect will not be present when they are distributed at no charge.

6.4.2.3 A variation on cap-and-trade: an emission reduction credit (ERC) system

Previous paragraphs have referred to a cap-and-trade permit system. A few comments are in order about an alternative variant of cap-and-trade, known as an emission reduction credit (ERC) system. In an ERC approach, a baseline profile of allowable emissions is established (for both aggregate emissions and emissions by individual sources that must sum to that aggregate). Emissions by any particular source above its baseline volume are subject to some prohibitive non-compliance penalty.[25] However, if a source emits less than its calculated baseline level, it earns a corresponding amount of emission reduction credits. Such credits can be sold to other sources that anticipate exceeding their baseline emission level.[26] The purchased ERCs constitute an entitlement to exceed baseline emissions without penalty. It is evident that in an ERC regime, each ERC is in principle equivalent to a marketable emissions permit. As such, one would expect that other things being equal the equilibrium market price of ERCs would be identical to that in a cap-and-trade regime.

ERC systems are sometimes used in association with pre-existing 'command and control' standards or licence schemes. Until recently, for example, US air and water quality standards control relied heavily on such hybrid schemes. The United States Environmental Protection Agency (US EPA) established national ambient air quality or permissible water pollutant concentration standards. To attain these standards, conventional command and control regulations consisting of required abatement technologies or ceiling on emissions flows were imposed on individual polluting sources. These have subsequently been augmented with the use of ERC systems.

But substantial efficiency gains may be available if some forms of tradability of licences is possible. An ERC system can bring this about. If any polluter succeeds in reducing emissions by a greater amount than is required by the standard it must satisfy, it obtains emission reduction credits of that quantity. The firm which acquires these emission reduction credits can engage in trades, selling some or all of its ERC to other firms, which then obtain a legal entitlement to emit pollutants beyond the standard which the USEPA imposed on them. Put another way, each firm was legally entitled to emit a quantity of pollutants up to the sum of its standard entitlement plus any ERC it has acquired.[27]

6.4.2.4 Flexible permit systems with offsets

The discussions we have taken you through above relating to cap-and-trade permits systems and the emissions reduction credit variant of cap-and-trade have implicitly worked on the assumption of a *closed system* being controlled. That is, the EPA

[25] Whether such penalties are sufficiently high so that, given the likelihood of detection, the expected cost of non-compliance exceeds the benefits of non-compliance, is an empirical matter. If such penalties are not of sufficiently high order, then the whole system will fail to meet its stated goals.

[26] If banking is allowed, they may also be used by the source at a later date.

[27] It is important to distinguish between generic features of any class of pollution control instrument and specific forms adopted at particular times and places. For example, USA ERC trading systems have incorporated other *distinctive* features that are not part of generic ERC regimes, such as an *offset policy* (which allows existing firms to expand, or new firms to enter, areas in which emission standards have not been met in the aggregate provided that they acquire sufficient quantities of ERC), and a *bubble policy* (which treats an aggregate of firms as one polluting source and requires that the bubble as a whole meets a specified standard).

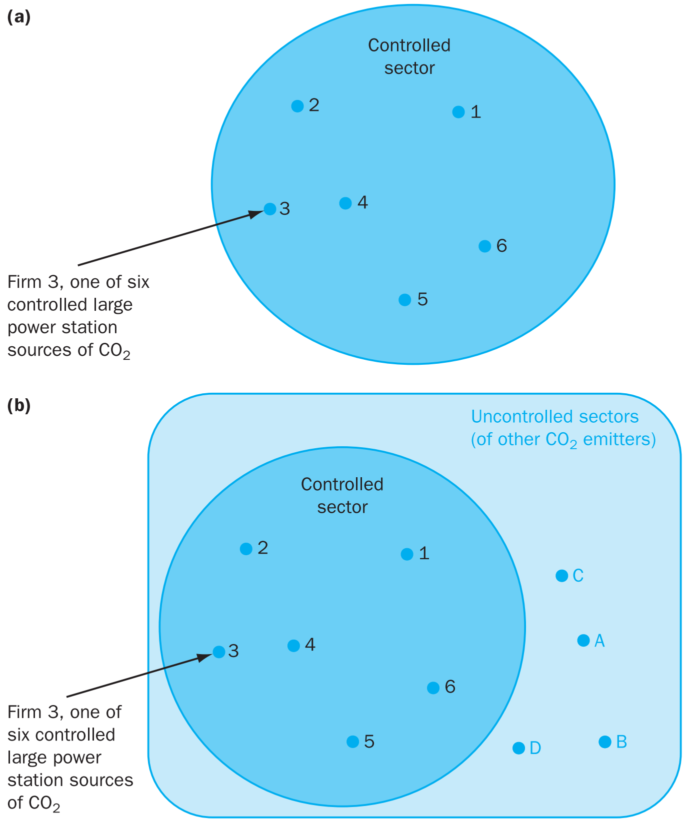


Figure 6.12 Two types of permit system. (a) A cap-and-trade permit system; (b) A flexible permit system with offsets

designates a carefully defined set of sources of a particular kind of polluting emission, and then imposes a cap on the total allowable quantity of that emission by all emission sources within the designated set. Permits, equal to that total allowable quantity, are then allocated to members of that set (by either free allocation or by auction). For example, the EPA might be concerned with capping CO_2 emissions (the particular kind of emission) by all fossil-fuelled large power generators (the designated set of emission sources).[28]

This is illustrated by the upper part of the schematic diagram in Figure 6.12, as Figure 6.12a. There, the controlled emission is CO_2, and the set of controlled sources consists of six large power stations.[29] Suppose that these six stations were emitting 100 million tons of CO_2 per period prior to the introduction of the permit system, and that the cap was set at three quarters of that level. Then 75 million tons of CO_2 emissions allowances per year are issued, and under a cap-and-trade system the EPA can be confident that no more than 75 million tons will be emitted by the controlled sources, provided of course that monitoring and enforcement is sufficiently efficacious. Moreover, if the set of controlled sources were defined to be *all* sources of CO_2 (almost

[28] Clearly it would be necessary to define 'large' in some appropriate and unambiguous way.

[29] The number six, chosen to keep our diagram manageable, is likely to be unrealistically small. For example, the European Union Emission Trading System (EU ETS), in its first two phases, covered more than 10 000 installations in the energy and industrial sectors which were collectively responsible for close to half of the EU's emissions of CO_2 and 40% of its total greenhouse gas emissions. With 25 EU member states, this amounts to an average of 400 per country (not all being power stations though).

certainly impossible in practice) then a cap-and-trade system would mean certainty in relation to *total* CO_2 emissions.

But now consider an alternative set of arrangements, illustrated schematically in the lower part of Figure 6.12 (as Figure 6.12b). Here the controlled set of emitters – again six large CO_2 sources – is an open sub-system within a larger system that also includes other, uncontrolled CO_2 sources. What is distinctive about this set of arrangements is the presence of offsets (or credits). Specifically, a controlled emitter is allowed to emit a greater amount of CO_2 than the quantity of permits it holds if it can make arrangements with an uncontrolled source to reduce its CO_2 emissions by a corresponding amount. These corresponding reductions by uncontrolled sources earn offsets for the controlled firms. They are then entitled to emit amounts equal to the sum of permits and offsets which they hold.

Why would a non-controlled organisation be willing to voluntarily reduce its emissions when the offsets accrue to a large controlled emitter? In general there will need to be a financial incentive for it to do so. One mechanism by which this can occur is if the controlled emitter pays for the CO_2 reduction. The controlled source will be willing to do so provided that the necessary payment for any given amount of emission reduction is smaller than the cost of purchasing the corresponding quantity of permits on the permits market.

This ability to make offset arrangements turns out to be the main advantage of the flexible permits with offsets system over pure cap and trade, for it allows a given total quantity of emissions reduction to be achieved at lower total cost. In other words, the flexibility creates greater cost-effectiveness. This follows from the fact that it can only be possible for a controlled firm to pay an 'outsider' a sum less than the equilibrium price in the permit market if emissions reduction has a lower marginal cost outside the controlled zone than inside the zone.

But there is one potentially serious disadvantage of this offsets system: the EPA may no longer be certain that net emissions are actually being reduced. Clearly, the offsets regime leads to the controlled firms emitting a greater amount than their total cap. If one knew for sure that CO_2 emission reductions taking place by uncontrolled organisations were genuinely additional – in the sense of being reductions which would not have taken place in the absence of this flexible permits regime – then there need be no worries on that score. Overall, system-wide, emission reductions as a result of this policy instrument will be equal to the magnitude of the difference between emissions before the policy instrument was chosen and the size of the cap.

But ensuring that offsets are only awarded when reductions are genuinely additional is extremely difficult to ensure. It requires that the EPA has an explicit projection of the future time paths of uncontrolled sources emissions under a 'business-as-usual' (BAU) or non-interventionist scenario. It also requires that the EPA is able to monitor the time paths of emissions of outsiders with whom offset arrangements are made, to compare these with the BAU paths, and to impose sufficiently strong deterrents to prevent spurious offset agreements from taking place. Whether these requirements can be met in practice is a moot point, but it seems likely that some non-additionality of emissions reduction will happen, so that overall emissions targets are overshot.

This discussion leads us to conclude that a choice between pure cap and trade and a flexible system with offsets involves an assessment of how to deal with the trade-off between greater certainty concerning the extent to which targets are actually meant and the lower cost per unit of emissions reduction that a flexibility system allows.

Some additional information on the complexities of marketable permit schemes that have been used in practice is given in Box 6.5. The examples considered there also include permit schemes in which what is being 'permitted' is something other than pollution emissions.

6.5 Pollution control where damages depend on location of the emissions

We now consider instruments designed to attain pollution stock (rather than emission flows) targets for non-uniformly-mixing stock pollutants (non-UMP). Chapter 5 explained why the spatial location of

Box 6.5 Marketable permits in practice

The use of marketable permits to attain environmental goals has become increasingly widespread in recent years, with new applications of marketable permits for conventional pollutants in nations such as Chile, China and Slovakia being noteworthy.

A principal use has been for emissions control. In the case of air emissions, applications have included sulphur dioxide (SO_2), ozone-depleting substances (ODS), nitrogen oxides (NO_X), lead in petrol, volatile organic compounds (VOC) and carbon dioxide (CO_2). Emissions into water and land systems covered by marketable permit schemes have included biological oxygen demand (BOD) and saline discharges into rivers. Common to all these applications has been the property that controlled emissions sources have been point (as opposed to diffuse) sources and relatively small in number. As marketable permit systems require the establishment of markets and careful monitoring of activity of market participants, these conditions are ones which are well suited to the use of this instrument.

There are also many examples of marketable permit schemes for purposes other than emissions control. Typically, these have concerned natural resource harvesting, exploitation, or use. Examples include the Australian system of water abstraction rights, construction or development permits for land management in the USA, France (housing in the Alps) and New Zealand (housing density), and permit systems for the harvesting of renewable resources (e.g. transferable fishing or logging quotas, several of which are described in Chapters 17 and 18).

An early indication of the promise that marketable permits schemes offered was the US lead trading scheme. Since the 1920s lead had been added to gasoline (petrol) by refiners to reduce premature combustion, allowing more powerful engines to be built. By the 1970s virtually all gasoline contained lead at an average of almost 2.4 grams per gallon, but at significant health cost in adult hypertension and cognitive development in children. As a result, in the mid-1970s USEPA acted to curtail lead use in gasoline by imposing increasingly tight standards on the lead content of gasoline. Some averaging was allowed over batches produced in any quarter, and later over different refineries owned by the same firm, implicitly allowing within-firm trading.

To facilitate the complete elimination of lead in fuel, from 1982 USEPA allowed between-firm trading and later banking (in effect, trading over time). Lead credits were created by refiners, importers and ethanol blenders (who reduced the lead content of gasoline by adding ethanol). Judged by market activity, the lead credit trading program was quite successful. Lead credit trading as a percentage of lead use rose above 40% by 1987. Some 60% of refineries participated in trading and 90% participated in banking by end of the program (Hahn and Hester, 1989a). The lead trading program is now regarded as a considerable success. The use of lead in leaded gasoline was sharply reduced over a short period of time without spikes in the price of gasoline that otherwise might have occurred (as with fixed standards on each batch of fuel, price would have been determined by highest marginal cost suppliers; but marginal cost differentials are eliminated by efficient trading of lead credits). Trading allowed the EPA to phase out the use of lead in gasoline much more rapidly than otherwise would have been feasible, as refiners faced very different opportunities for reducing the lead content of gasoline.

In 1992, Cropper and Oates (1992, pp. 729, 730) had written:

> . . . marketable permit programs are few in number and often bear only a modest resemblance to the pure programs of economic incentives supported by economists. . . . As we move into the 1990's, the general political and policy setting is one that is genuinely receptive to market approaches to solving our social problems. Not only in the United States but in other countries as well, the prevailing atmosphere is a conservative one with a strong predisposition towards the use of market incentives wherever possible, for the attainment of our social objectives.

The prescience of this is borne out by the enormous growth in the use of marketable emissions permit programmes in the following 25 years, particularly in the USA, to a lesser degree in EU countries, and internationally under the flexible mechanisms for greenhouse gas control under the Kyoto Protocol. Details of early implementations of marketable permit programmes can be found in surveys by Cropper and Oates (1992), Hahn (1989, 1995), Hahn and Hester (1989a, 1989b), and Opschoor and Vos (1989). More recent initiatives are surveyed in

Box 6.5 ***continued***

EPA (2001, 2004). Important catalysts to this growth have been the 1990 amendments to the Clean Air Act in the United States, and widespread ratification of the Kyoto Protocol, initiated at Kyoto, Japan in 1997 (and to be discussed at length in Chapter 9). The industrialised countries, in agreeing to a programme of greenhouse gas emissions limits, implicitly agreed that the rights to emit pollutants could be traded not only within but also between nations.

We now take the reader briefly through the two most well-known marketable permit programmes: the USA SO_2 and NO_X programmes operated by United States Environmental Protection Agency, EPA; and the European Union Emissions Trading Scheme, EU ETS.

The EPA's marketable permits programmes (in conjunction with related schemes) are designed to achieve substantial and cost-effective reductions in emissions of sulfur dioxide (SO_2) and nitrogen oxides (NO_x). The programmes employ both traditional and market-based instruments, the latter being looked at here.

Sulphur dioxide (SO_2) reductions

For SO_2, the Clean Air Act set a goal of reducing annual emissions by 10 million tons below 1980 levels. In Phase II of the program (in effect in 2009), the law placed controls on all existing fossil fuel-fired power plants with an output capacity of greater than 25 megawatts, and new utility units, located in eastern and midwestern states. These states are those in which acid rain problems were most acute. (Similar schemes also operate in other individual states or collections of states.)

Under this system, affected utility units are allocated allowances based on their historic fuel consumption and a specific emissions rate. A permanent ceiling (or cap) of 8.95 million allowances (in units of tons of SO_2) for total annual allowance allocations to controlled utilities. Although allowances (permits) are issued on this free grandfathering basis, and once issued can be traded through organised permit markets, EPA holds an allowance auction annually. The auctions help to send the market an allowance price signal, as well as furnish utilities with an additional avenue for purchasing needed allowances. The regime also includes an Opt-in Program for additional sulfur dioxide (SO_2) emitting sources; this allows sources not required to participate in the Acid Rain Program the opportunity to enter the program on a voluntary basis and receive their own SO_2 allowances. By reducing emissions below its allowance allocation, an opt-in source will have unused allowances, which it can sell in the SO_2 allowance market. Opting in will be profitable if the revenue from the sale of allowances exceeds the combined cost of the emissions reduction and the cost of participating in the Opt-in Program. This should reduce the cost of achieving the 10 million ton reduction in SO_2 emissions mandated under the Clean Air Act because participating sources that reduce their SO_2 emissions at a relatively low cost can transfer their allowances to controlled electric utilities where emission reductions are more expensive. Finally, EPA has set aside a reserve of 300 000 allowances; those utilities that either implement demand-side energy conservation programs to curtail emissions or install renewable energy generation facilities may be eligible to receive bonus allowances from this reserve.

Nitrogen oxides (NO_x) reductions

Nitrogen oxides are important precursors of ground-level ozone (smog). Controls have the objectives of reducing the emissions of these substances, and minimising the transport of ozone across large distances. The Clean Air Act Amendments of 1990 set a goal of reducing NO_x by 2 million tons from 1980 levels by 2000. The program, started in 1996, focuses on one set of NO_x emitters, coal-fired electric utility boilers. A substantial part of the way in which EPA regulates NO_x emissions is through a requirement that coal-fired utility boilers are required to install low NO_x burner technologies and to meet tight emissions standards. These required-technology command-and-control instruments have been hugely effective in reducing emissions.

The NO_x program embodies many of the same principles of the SO_2 trading program, but it does not 'cap' NO_x emissions as the SO_2 program does. It also utilises a very different form of trading system. There are two options for compliance with the emission limitations:

- compliance with an individual emission rate for a boiler;
- averaging of emission rates over two or more units to meet an overall emission rate limitation.

Box 6.5 ***continued***

In 2003, EPA began to administer a set of somewhat diverse pre-existing NO_x trading schemes, introducing the 'NO_x Budget Trading Program' (often referred to as under the 'NO_x SIP Call'). This is a market-based cap and trade program to reduce emissions of nitrogen oxides from power plants and other large combustion sources in the eastern United States. NOx is a prime ingredient in the formation of ground-level ozone, a pervasive air pollution problem in many areas of the eastern United States. The NO_x Budget Trading Program was designed to reduce NO_x emissions during the warm summer months, referred to as the ozone season, when ground-level ozone concentrations are highest. For those states that elect to participate in the program, EPA provides a template 'NO_x Budget Trading Program rule (Part 96)'. This provides sources with a complete trading program including provisions for applicability, allocations, monitoring, banking, penalties, trading protocols and program administration. States choosing to participate in the NO_x Budget Trading Program have the flexibility to modify certain provisions within the model rule.

General features of US EPA Air Emissions Programs

Each controlled source must continuously measure and record its emissions of SO_2, NO_x, and CO_2, as well as heat input, volumetric flow, and opacity. In most cases, a continuous emission monitoring (CEM) system must be used. Sources report hourly emissions data to EPA on a quarterly basis. The emissions monitoring and reporting systems are critical to the program. If annual emissions exceed the number of allowances held, the owners or operators of delinquent units must pay a penalty of $2000 (adjusted for inflation) per excess ton of SO_2 or NO_x emissions. In addition, violating utilities must offset the excess SO_2 emissions with allowances in an amount equivalent to the excess.

The General Accounting Office recently estimated that the allowance trading system could save as much as $3 billion per year – over 50% – compared with a command and control approach typical of previous environmental protection programs.

We also note that marketable permit regimes in the USA cover more than the pollutants described above. For example, the Clean Air Mercury Rule (CAMR) aims to reduce mercury emissions from coal-fired power plants through 'standards of performance' for new and existing utilities and a market-based cap-and-trade program.

Sources: Tietenberg (1990), Goodstein (1995), OECD (1999), Hahn and Hester (1989a), Anderson *et al.* (1990), Kerr (1993), Kerr and Mare (1995). Material on lead trading is taken from the US EPA 'National Center for Environmental Economics' website at http://yosemite.epa.gov/ee/Epalib/incent.nsf/c484aff385a753cd85256c2c0057ce35/4a0870c300d11b32852564f4004eaa17!OpenDocument

emissions is of importance in this case. It will be convenient to deal with the particular example used there of air pollution in an 'airshed' that contains several spatially distinct receptor areas and many emission sources. However, our results will apply, with suitable change of terminology, to any non-UMP.

One way in which the EPA may handle these issues is by controlling *ex ante* the location of polluters and people affected by pollution. Indeed, in the very long run, the best way of dealing with this problem is likely to be zoning: prohibiting new sources from being set up in, or near to, the airshed, and requiring existing sources to move away from the receptor areas. But what should the EPA do when the location of polluters and people is already determined, and moving either is not a feasible option?

When the location of sources is regarded as being fixed, pollution control must work by regulating in some way the emissions from those sources so as to meet the relevant air quality standards.[30] Once again, it is assumed here that standards have already been set. Those standards will consist of maximum allowable concentration rates of the stock pollutant in

[30] The terms 'targets' and 'standards' are being used synonymously here.

each of the relevant receptor areas. These standards may be 'efficient' targets (those we analysed in Chapter 5) or they may not. To the authors' knowledge, no operative targets for non-UMP have ever been set in terms of economic efficiency. So it will be sensible to deal with the case of arbitrary specific targets. We assume that in pursuit of its objectives the EPA seeks to reach targets at least cost.

Our exposition proceeds as follows. We first establish a benchmark of what is required in order to obtain the maximum allowable emissions from each source that meets the two constraints: (i) the pollution target is reached in every receptor area, *and* (ii) at minimum possible overall cost. Having established that benchmark, we then consider how well each of the following three instruments perform against that benchmark:

1. non-transferable emissions licences allocated to each source (a command and control approach);
2. emissions taxes or emissions abatement subsidies;
3. marketable emissions permits.

6.5.1 Obtaining the benchmark cost-minimising solution

The benchmark we are seeking is the solution to the problem of calculating the maximum allowable emissions from each source so that the pollution target is reached in every receptor area, *and* at minimum possible overall cost. That is, a cost-minimisation problem needs to be solved. Appendix 6.1, available on the Companion Website, goes through the maths of this cost-minimisation problem and obtains general, analytical results. Here we just indicate the way in which the problem is set up, and interpret the main results. An Excel workbook, *Ambient instruments.xls*, also available on the Companion website, provides a worked numerical example of the problem being investigated.

It will be convenient to recap the notation we use for non-UMP. The airshed being considered contains J spatially distinct pollution receptors (indexed $j = 1, 2, \ldots, J$) and N distinct pollution sources (indexed $i = 1, 2, \ldots, N$). The transfer coefficient d_{ji} describes the impact on pollutant concentration from source i in receptor j. Pollution at location j, A_j, is the sum of the contributions to pollution at that location from all N emission sources:

$$A_j = \sum_{i=1}^{N} d_{ji} M_i \tag{6.1}$$

where M_i is emissions from source i. Section 5.6 provided much of the theoretical background for the case of non-UMP, but that section was concerned with the choice of efficient emissions target. Here our interest is not in target choice but rather in instrument choice, and so we assume that targets (for pollutant stocks in each receptor area) have already been set. As far as licences are concerned, our task is to find the level of emissions from each source that minimises the overall cost of reaching those targets. For tax (subsidy) instruments, our goal is to find the tax (subsidy) rate or rates that will reach those targets at least cost. We shall also be interested in how a marketable permit system could be designed in this case.

Let A_j^* denote the EPA's target pollutant concentration at receptor j. (The symbol A can be thought of as ambient air quality, another expression for the concentration rate of some relevant air pollutant.) For simplicity we suppose that the target for each receptor area is the same, so that $A_j^* = A^*$ for all j.[31] The overall goal of the EPA is that in no area should the pollutant concentration exceed A^*. That is,

$$A_j = \sum_{i=1}^{N} d_{ji} M_i \leq A^* \quad \text{for } j = 1, \ldots, J \tag{6.2}$$

Next suppose that the EPA adopts one single criterion in pursuing its objective. It wishes to achieve the overall target (given in equation 6.2) at least cost. The solution (as we show in Part 8 of Appendix 6.1) requires that

$$\mathrm{MC}_i = \mu_1^* d_{1i} + \mu_2^* d_{2i} + \ldots + \mu_J^* d_{Ji}, \; i = 1, 2, \ldots, N \tag{6.3}$$

[31] Appendix 6.1 considers the more general case in which targets differ between each receptor area. It obtains the special case (being considered here) of identical receptor stock or concentration targets as a special case of that more general result.

where MC_i denotes the marginal abatement cost of firm i. We shall interpret equation 6.3 in a moment. Meanwhile, note that the systems 6.2 and 6.3 constitute $N + J$ equations which can be solved for the cost-minimising values of the $N + J$ unknowns (N emissions levels and J shadow prices, $\mu_j^*, j = 1, \ldots, J$). It is best to leave discussion of the *properties* of such a solution to the following three sections, which describe how this would be attained in each of the three instrument cases we are considering. So we turn next to that task.

6.5.2 Using non-transferable emissions licences

The use of non-transferable emissions licences is simple in principle. All that is required is for the EPA to solve the cost-minimisation problem described above, and then to allocate licences to each source in the quantities that emerge from the solution to that problem. These licence quantities should be the N solution values of M_i^*. Note that even if firms have identical marginal abatement cost functions, they will *not* do equal amounts of emission abatement. This can be seen from the fact that the transfer coefficients on the right-hand side of 6.3 will vary from firm to firm. Hence the value of the whole expression on the right-hand side of 6.3 will differ between firms, and so their marginal abatement costs must differ too. That implies doing different amounts of abatement.

This may be compared with the condition that we found earlier for a uniformly mixing pollutant,

$$MC_i = \mu^*, \ i = 1, 2, \ldots, N$$

which means that the marginal cost of emissions abatement is equal over all pollution sources. Hence, if firms had identical abatement cost functions they would do identical amounts of abatement. The intuition behind the result that firms will abate to different amounts where they emit non-UMP is simple. Emissions from some sources have more damaging consequences than emissions from others, because of the way in which emissions become distributed over the area of concern. Those sources whose emissions lead to relatively high damage should have relatively low emissions (and so relatively high marginal abatement costs).

It is noticeable that the command and control approach can only be used cost-efficiently if the full solution to the cost-minimising solution is known. In particular, the EPA must be able to compute the N solution values of M_i^* so that it can distribute licences accordingly. This is, at the very least, a daunting task, and one which seems hard to implement. It has already been established that for simple flow pollutants and for uniformly mixing stock pollutants the use of tax or subsidy instruments, or marketable permit instruments, did not require knowledge of each firm's marginal cost function. (It did not require that information as it could assume that in an efficient allocation, all marginal costs would be equal, and that could be guaranteed by a uniform tax or by a single market in tradable permits.) Let us now see whether this advantage of incentive-based instruments over command and control carries over to the case of non-UMP.

6.5.3 Using emissions taxes or emissions abatement subsidies

We now turn to consider a tax (or subsidy) instrument. This requires a bit more care in interpreting equation system 6.3. The μ_j^* terms that appear in each of the N equations are shadow prices. There is one of these for each receptor area. Each denotes the monetary value of a worsening of the pollution stock by one unit in that area. The d_{ij} coefficients tell us how many units pollution increases by in receptor j if emissions from source i rise by one unit. So for example $\mu_2^* d_{2i}$ gives the monetary value of damage that accrues in area 2 from an additional unit of emissions in source i. By summing these values over all source areas (that is, $\mu_1^* d_{1i} + \mu_2^* d_{2i} + \ldots \mu_J^* d_{Ji}$) we find the total value of damage caused in all receptor areas by an additional unit of emission from i. Cost-efficiency requires that each firm pays a tax on each unit of emission, t_i, (or receives a subsidy on each unit abated, s_i) equal to the value of that damage, so we have

$$t_i = s_i = \mu_1^* d_{1i} + \mu_2^* d_{2i} + \ldots + \mu_J^* d_{Ji}$$

Note that the tax (subsidy) rate will now *not* be the same for each firm. This is just what we would expect for non-UMP as damage varies according to the location of emission source.

There is one important corollary of this. As tax or subsidy instruments require that rates are unique to each pollution source, one of the attractive features of these instruments (that a single rate can be applied over all polluters) no longer applies. Indeed, a single tax rate would *not* lead to a cost-effective abatement programme in this case.

If the EPA were determined to use a tax instrument, nonetheless, and tried to calculate the source-specific tax rates, it would require exactly the same amount of information as a command and control system does. In particular, it would need to know the marginal abatement cost function for every firm. Hence a second desirable property of a tax instrument – that it does not need knowledge of an individual firm's costs – also disappears. All in all, one would expect much less use to be made of pollution tax or subsidy instruments in the case of non-uniformly-mixing air, water or ground pollution than with a uniformly mixing pollutant.

6.5.4 Using marketable emissions permits

How would marketable permits work in this case? The system – known as an ambient marketable permits or spatially differentiated system – would operate as follows:

1. Each receptor site will have a pollution concentration target. As before, we assume that this is the same for all receptors, A^*.
2. For each receptor site, the EPA must calculate how many units of emission can be allowed to arrive at that site before the pollution target is breached. More formally, it must calculate how many 'emissions permits' there can be that will allow firms to decrement (that is, worsen) ambient concentrations at that site.
3. These permits are issued to pollution sources, either by competitive auction or by free initial allocation ('grandfathering' if this is done proportionally to previous unregulated emission levels).
4. A pollution source is prohibited from making an emission to any receptor site above the quantity of permits it holds for emissions to that site. Each firm will, therefore, be required to hold a portfolio of permits to worsen concentrations at specific receptor areas.
5. A market for permits will emerge *for each receptor area*. Each polluting source will trade in many of these markets simultaneously. The results of these trades will determine a unique equilibrium price in each market.
6. Permits for each receptor area are freely marketable on a one-to-one basis, but this does not apply to permits for different receptors.

Note that 'emissions permits' have a special meaning in this context. They are not unrestricted rights to emit. Rather, they are rights to emit such that pollutant concentrations will worsen by a particular amount at a particular place. So, for example, if I want to emit one unit, and that will worsen pollution by 3 units at receptor 1 and by 4 units at receptor 2, I must buy a permit to worsen pollution (by those amounts) in each of the two markets.

How does this relate to equation 6.3? The J shadow prices μ_j^* correspond to the equilibrium permit prices in each market. At the least-cost solution, a firm will equate the marginal cost of emissions abatement with the marginal cost of not abating the emission. The right-hand side of equation 6.3 gives this latter cost, which is a weighted sum of these permit prices. The weights attached to the permit price for receptor j will be the impact that one unit of emissions has on pollutant concentration at site j. Thus the right-hand side gives the cost to the firm, in permit prices paid, for one unit of its emissions.

Clearly, the administration of an ideal least-cost marketable permit system is hugely demanding. However, it does have one major advantage over both command and control and tax/subsidy instruments: the EPA does not have to know the marginal abatement cost function of each firm in order to achieve the pollution targets at least cost. This is the major reason why emissions permits have attracted so much attention from economists, and why they are being introduced in practice in a form similar to that outlined above.

There are as yet no actual examples of systems that match this ideal form exactly. Existing permit systems are only approximations to the ideal type. The most important departure in practice is the absence of separate markets for permits for each receptor. (Systems in practice tend, instead, to have markets for each type of pollution generator.) You should be able to see that the absence of separate receptor markets may substantially increase the true cost of achieving pollution targets.

The extent to which an ideal least-cost marketable permit scheme would attain ambient standards at lower cost than some alternative instruments has been analysed by several authors using simulation techniques. We outline one of these studies (Krupnick, 1986) in Box 6.6. Krupnick's study also highlights another matter of considerable importance: abatement costs can rise very sharply as the desired targets are progressively tightened.

Box 6.6 Costs of alternative policies for the control of nitrogen dioxide in Baltimore

Nitrogen dioxide (NO_2) is a good example of a non-uniformly-mixing pollutant. Alan Krupnick (1986) investigated the cost of meeting alternative one-hour NO_2 standards in the Baltimore area of the United States. He compared a variety of control programmes applied to 200 large emission point sources in the area. He identified 404 separate receptor areas in the region. Krupnick considered three alternative standards applied for each receptor area: 250, 375 and 500 $\mu g/m^3$ control.

Simulation techniques are used to estimate total abatement costs for each of several different policy instruments. We deal here with four of the cases that Krupnick investigated:

- the least-cost instrument: a spatially differentiated ambient-pollution marketable permits scheme of the type discussed in the text;
- a type-specific fee: an effluent charge with charges differentiated by source type (but not by receptor areas impacted);
- a uniform fee: an effluent charge not differentiated by source type (nor location of impact);
- a hybrid instrument, labelled RACT/least-cost: a mixture of command and control and incentive instruments. The RACT part takes the form of a technology standard ('Reasonably Available Control Technology') which is imposed on all firms. For firms that fail to meet (weaker) national air-quality standards, market incentives are used to induce further emissions reductions (the least-cost part).

The results of Krupnick's simulations (for two ambient targets) are shown in Table 6.5. Numbers not in parentheses refer to the stricter target of 250 $\mu g/m^3$, those in parentheses the weaker target of 500 $\mu g/m^3$. These targets were selected in view of the fact that uncontrolled emissions led to high ambient pollution levels of around 700–800 $\mu g/m^3$ at several receptor sites, and technology studies suggest that targets stricter than around 190 $\mu g/m^3$ are unobtainable given the presence of the existing point sources.

Table 6.5 Simulation results for the cost of meeting two ambient targets

	Emissions reduction (%)		Abatement costs $US millions/year	
Least cost (ambient permits)	32	(6)	1.663	(0.066)
Type-specific fee	34	(6)	1.719	(0.066)
RACT/least cost	42	(36)	2.200	(1.521)
Uniform fee	73	(21)	14.423	(0.224)

Source: Adapted from Krupnik (1986), Tables II and III

Comparing first the costs of attaining different targets, Krupnick notes that 'compliance costs rise steeply as the standard is tightened, regardless of the policy simulated. In the least-cost case, costs rise by a factor of 25 (from \$66 000 to \$1.633 million) when standards are halved (from 500 to 250 $\mu g/m^3$.' The smaller proportionate increase in the hybrid case (RACT/least-cost) is due to the fact that the technology controls imposed by RACT give the firms little additional room for manoeuvre for further cost reductions when the standard is made stricter.

Notice that the emissions reduction is relatively small for the least-cost control compared with others. This happens because the target being sought is not a given total emissions reduction but a maximum ambient pollution standard over the whole area. Several of the instruments are inefficient (in abatement cost terms) because they operate in a more uniform manner than the spatially differentiated least-cost permit method. In so doing, the optimal distribution of abatement effort is not being applied, and excessive amounts of control are being adopted on many pollution sources.

For the type-specific fee, control costs are not much larger than for the least-cost method (and are identical for the weaker control). A fee that distinguishes between different types of polluter

Box 6.6 *continued*

does seem able to mimic fairly well a proper spatially differentiated permit (or tax) approach. This is reassuring, as type-specific fees are likely to be used in practice instead of least-cost ambient permit methods as a result of their much greater simplicity. In contrast, note that when a uniform fee is imposed to achieve the stricter ambient standard (and where uniformity means that no effort is made to relate the charge to impact of emissions on ambient levels at various places) control costs increase very dramatically. A uniform fee can result in the largest emission reduction, but without doing any better in terms of ambient standards, and at hugely additional cost. Note, finally, that a single market emissions permit system would have an identical effect to that of a uniform fee. Spatially differentiating permit markets offers huge cost savings in principle.

6.6 A comparison of the relative advantages of command and control, emissions tax, emission abatement subsidy and marketable permit instruments

In this section, we bring together a set of results obtained earlier in the chapter, and introduce a few additional results, in order to assess the relative merits of alternative pollution control instruments.

6.6.1 Cost-efficiency

For a flow pollutant or for a uniformly-mixing stock pollutant, an emissions tax, or an emissions abatement subsidy or a marketable permit system can achieve any emissions target at least cost. A command and control (CAC) regulation instrument may, but will not usually, be cost-efficient. In order to be cost-efficient, the EPA must know each polluter's marginal cost of abatement function so that an emission control can be calculated for each firm that will equalise marginal abatement costs. It is very unlikely that this requirement will be met. Hence for a flow pollutant or for a uniformly-mixing stock pollutant a command and control quantity regulation approach is inefficient relative to a tax, subsidy or marketable permit scheme, and so will achieve any specified target at a higher real cost. This result is now widely agreed, and simulation evidence suggests that command and control instruments can, in some circumstances, be orders of magnitude more costly than least cost solutions from well-designed incentive-based instruments.

Three arguments underlie the tenet that market-based incentive approaches are likely to be more efficient than regulation and control. First, markets are effective in processing information; second, market instruments tend to result in pollution control being undertaken where that control is least costly in real terms; and third, market-based approaches generate dynamic gains through responses over time to their patterns of incentives. We discuss this last point further below.

However, stringent conditions are necessary for markets to guarantee efficient outcomes. Policy instrument choice takes place in a 'second-best' world, where results are much less clear. The absence of markets (including those for externalities and public goods), asymmetric information, moral hazard and other instances of market failure all point to possible benefits of command and control-based public intervention or to the inappropriateness of complete reliance on markets and market instruments in particular circumstances. (See Fisher and Rothkopf, 1989, for an excellent survey.) In the previous edition of this textbook, some empirical evidence on these propositions was presented (in Box 7.10 of Chapter 7, 3rd edition). That box has not been reproduced in this edition, but remains available on the Companion Website should you wish to consult it.

For a non-UMP stock pollutant, the remarks above need to be qualified. Cost-effective command and control systems, as before, require knowledge of individual firms' marginal cost of abatement functions. But so too do tax and subsidy instruments. In general, only transferable permit schemes do not require that knowledge. This gives permit systems substantial potential advantages over others in cases where pollution stocks do not uniformly mix.

6.6.2 Monitoring, administering and enforcing compliance costs

Little or nothing has been said so far about the costs associated with monitoring, administering and enforcing compliance for each instrument. Yet these costs could be quite substantial. If they are large, and if they differ significantly between instruments, these costs are likely to have an important bearing on which type of instrument is least-cost for achieving some target. One reason for the prevalence of minimum technology requirements as a pollution control instrument may be that these costs can be low relative to those of instruments that try to regulate emissions output levels.

6.6.3 Long-run effects

From the point of view of the EPA, instrument selection will depend on the degree to which the amount of pollution control varies with the passage of time for any particular instrument. An important consideration concerns whether or not the long-run effect is markedly different from the short-run effect, and if so, whether it weakens or strengthens. The long-run effect of an instrument depends mainly on two things: net income effects and technological innovation effects. We consider each of these in turn.

Net income effects

Changes in net income arising from the operation of a pollution control instrument can affect the long-run industry size. We noted earlier that subsidy schemes may have the (environmentally) undesirable property of increasing the long-run size of the targeted industry through positive income effects. Similar issues were raised in comparing alternative methods of initially allocating marketable permits.

It is possible in principle to design control regimes that are revenue-neutral. For example, firms in a subsidised industry may be required to make lump-sum payments which sum to the total value of subsidies. This would preserve the incentive effects of subsidy systems without allowing long-run effects arising from income changes. However, it may be politically difficult to implement such a scheme, and there may be reasons why government does not wish to match receipts and payments in such a way.

Technology effects

A second route through which long-run effects may transmit is via induced impacts on the rate of technological innovation. There are two aspects to this. One route concerns what are sometimes called *dynamic efficiency* effects. These arise from the pattern of innovation incentives generated by a pollution control instrument. A common argument in this regard is that command and control instruments have poor long-run properties because they generate weak incentives for innovation (see, for example, Jaffe and Stavins, 1994). The binary nature of many such instruments (you reach the target or you do not reach it) creates a discrete switch in behaviour: once a required target has been obtained there is no longer any incentive to go further.

In contrast, an emissions tax (or abatement subsidy) is likely to generate a dynamically efficient pattern of incentives on corporate (and consumer) behaviour. The incentive structure operates to continually reward successful environmentally friendly innovation. In a market-based scheme, every unit of emissions reduction is rewarded by a tax saving. The key issue here is what incentives firms face in developing pollution-saving technology or developing new, environmentally cleaner products. Under a emissions tax scheme, these incentives may be strong, as we show in Figure 6.13.

Area Ω is the saving that would result if marginal costs were lowered from MC_1 to MC_2 and the emissions level was unchanged. But if marginal cost were lowered in this way, the firm's profit-maximising emissions abatement level would rise from Z_1^* to Z_2^*, and so an additional saving of $\wedge$ would accrue to the firm. The firm has an incentive to develop new technology to abate emission if the total costs of developing and applying the technology are less than the present value of the savings $\Omega + \wedge$ accumulated over the life of the firm.[32] In contrast, in a command and

[32] Note that the optimal tax rate would change as new technology lowers control costs, so matters are a little more complicated.

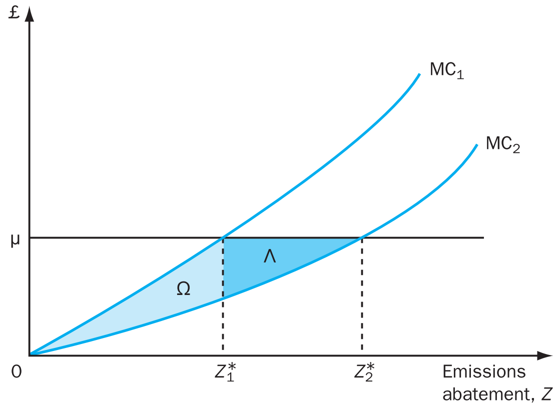


Figure 6.13 Dynamic incentives under emissions tax controls

control (CAC) regulatory system, dynamic incentives are weaker or non-existent. As we said above, if a target is set in (non-marketable) quantitative terms, then once that target has been met there is little or no further incentive on the polluter to reduce emissions.

But there is a second aspect that weakens the force of these arguments. Some researchers believe that technological change can be driven from above. Suppose that the EPA identifies best-practice environmentally friendly technology, and imposes this as a requirement on firms through minimum acceptable technology regulations. Not only will this have a direct effect on spreading technology diffusion, but the indirect effects may be powerful too. Barriers due to frictions, lack of information, and other market imperfections that may lead firms to be over-cautious or unable to act voluntarily no longer bite in the face of imposed requirements. Moreover, these changes have catalytic effects which set in motion spurts of innovation as learning effects occur. These kinds of arguments are likely to have most relevance for technological innovation and diffusion in developing economies.

It is difficult to arrive at unequivocal conclusions from all this. However, a reasonable conclusion must be that, in some circumstances at least, technology-based controls and other command and control instruments will have superior long-run properties to market-based instruments.

6.6.4 Double dividend

In the previous chapter, we noted the possibility that some environmental regulation schemes may generate a so-called double dividend. It seems likely that the availability and size of a double dividend will vary from one circumstance to another, and on which instrument is being used. A sensible choice of instrument should consider these matters.

6.6.5 Distributional impacts and fairness

Finally, we note that the distributional consequences of a pollution control policy instrument will be very important in determining which instruments are selected in practice. Different instruments for pollution control have different implications for the distribution of income within an economy. We have already examined the direct business financial gains and losses (which are, of course, exactly mirrored by offsetting government financial losses or gains). It is also necessary to think about the consequences for income and wealth distribution in society as a whole. For example, an emissions tax imposed upon fossil fuels will indirectly affect final consumers who purchase goods that have large energy input. Individuals for whom heating comprises a large proportion of their budget may well experience quite large falls in real income. Indeed, many kinds of 'green taxes' are likely to have regressive effects upon income distribution.

It is important to distinguish between income shifts that are merely redistributive and do not correspond to any real resource gains and losses to the economy, and real income changes which do imply real resource changes for the economy as a whole. The latter arise because pollution control does involve real costs. Of course, by having less pollution, there are benefits to be had as well, which in a well-designed pollution control programme should outweigh these real costs. Nevertheless, the beneficiaries and losers will not be the same individuals, and it is this that one is concerned with when discussing the equity or fairness of an instrument.

It should also be noted that emissions taxes (and other environmental controls) have important implications for the relative competitiveness of national economies. (See Chapter 10 for more on this.) Some analysts have advocated a switch from taxes on labour and capital to taxes on emissions to avoid excessive tax burdens, and schemes have been proposed to penalise nations that attempt to gain competitive advantage by not introducing emissions taxes. Good discussions of these issues are to be found in Bertram *et al*. (1989), Brown (1989), Grubb (1989a), Hansen (1990), Kosmo (1989) and Weizşker (1989).

As we noted earlier, where a particular instrument has an adverse financial effect on one sector of the economy, it is open to the government to use compensating fiscal changes to offset those changes so that the distribution of income and wealth between individuals is not systematically changed. For example, the financial transfers implied by an emissions tax scheme could be compensated for by lump-sum payments to firms or by abatement subsidy payments. And losses by poorer groups facing higher energy bills, for example, could be compensated for by other fiscal changes.

The main point here is that additional tax revenues received by government could be distributed to groups adversely affected by the initial policy change. However, the difficulties in designing distributionally neutral packages are immense. Where compensation is paid to individuals or groups for whom the tax incidence is considered excessive, the form of compensation should be designed not to alter behaviour, otherwise the efficiency properties of the instrument will be adversely affected. This implies that lump-sum compensation should be used where possible. Compensation schemes of this form rarely happen in practice. Nevertheless, decision makers do have this option; whether they choose to exercise it is another matter.

Box 6.7 Economic instruments to reduce waste and encourage recycling

Municipal and other waste charges

Bressers and Schuddeboom's (1994) survey of economic instruments reports that municipal waste user charges are levied in 18 of the 21 industrialised countries (all but New Zealand, Portugal, and the United Kingdom) that it surveyed. The charges are usually (but not always) flat rates for households and unit rates for commercial generators. Note that unit rates are likely to have a greater incentive effect than flat rates that are independent of quantities of waste generated. The charges usually fund waste collection and/or disposal, but in many countries are also set high enough to exert an incentive effect to reduce waste.

Denmark, for example, levies the highest fee on waste delivered to landfills and a somewhat lower fee on waste delivered to incineration facilities. Recycled wastes are not charged. Since the charges were introduced in 1987, the quantity of waste registered at disposal facilities has dropped and the reuse of building waste as filling material for road construction and other purposes has increased.

In the Netherlands, a charge equivalent to approximately $18 per ton on landfill disposal of waste came into effect on 1 January 1995 as part of a broader environmental tax law. The main purpose of the charge is to raise revenue for the national budget, but a secondary purpose is to discourage waste generation. To promote incineration as a disposal method, incineration is exempt from the tax. The size of the charge relative to the average waste treatment costs of 82 Dfl ($50) per metric ton suggests that the tax could have significant incentive impact.

The United Kingdom landfill tax, which dates from 1 October 1996, is managed by Entrust. The tax rate was £2 ($3) per metric ton for inactive waste such as bricks and £7 ($10.7) per metric ton for other waste. Landfill operators pay the tax and can raise disposal fees to recover their tax payments. The British Customs and Excise office said that the tax is 'designed to use market forces to protect the environment by making the disposal of waste in landfill sites more expensive'. Businesses' national insurance contributions were cut to compensate for the effect of the tax on business. A tax credit scheme allows landfill operators to donate up to 6.5% of their landfill tax liability to environmental projects in return for a 90% tax credit.

The tax is designed so that incentives exist to reduce waste flows. However, since its inception, operation of the tax has been plagued by concerns that waste has been disposed of illegally to avoid landfill tax charges. This illustrates the point that incentive-based instruments for environmental control may be ineffective unless there is careful monitoring and methods for ensuring compliance.

Outside the OECD, South Korea introduced a system in 1995 under which household waste can be disposed of only in standardised bags sold in officially designated places. Bag prices

Box 6.7 *continued*

in the metropolitan areas of the capital city of Seoul range from 60–80 won ($0.08–0.10) for five-litre bags to 1090–1450 won ($1.41–$1.88) for 100-litre bags. Prices are determined by local governments and vary slightly from area to area. The amount of waste sent to landfills was approximately 40% lower for the six months after implementation of the system. Unfortunately, a large quantity of the decrease was attributable not to waste reduction or recycling, but rather to uncontrolled incineration or private disposal. The plastic bags themselves are not biodegradable and thus pose disposal problems; moreover, the bag fees are too low to cover waste disposal costs.

In 1994, Turkey introduced an Environmental Cleanup Tax on waste to raise revenue and to discourage waste generation. The monthly rate was set at 25 000–100 000 TL ($0.37–$1.47) for households and 25 000–5 000 000 TL ($0.37–$295) for other generators. The Cleanup Tax was also imposed on wastewater.

Australia, Austria, Belgium, Finland, France, and several German states impose charges on hazardous waste disposal. Austria's tax of 200 S ($19) per metric ton is used to fund the clean-up of contaminated land. France has imposed a tax on the disposal of 'special industrial wastes', a category including asbestos, chrome, lead, solvents, and other specified substances. The tax rose from 20 F ($4) per metric ton in 1994 to 40 F ($8) per metric ton in 1998.

The Netherlands and the Flanders region of Belgium impose charges on animal manure disposal to limit soil pollution. In the Netherlands, individuals are permitted to dump the manure equivalent of 125 kg of phosphate per hectare per year free of charge. Quantities between 125 and 200 kg are subject to a charge of 0.25 Dfl ($0.15) for every kg over 125 kg, and quantities over 200 kg to a charge of 0.5 Dfl ($0.3) per kg.

Waste charges have also been levied in a number of less industrialised countries, including the Czech Republic, China, Estonia, Hungary, Poland, Russia, and the Slovak Republic. Municipal waste charges for households and businesses in the Czech Republic, which have been in place since before World War II, were significantly increased in 1992. Municipalities determine prices. One problem with the increased charges is that they appear to have led to an increase in illegal dumping.

Since 1992, the Czech Republic has also levied two types of charges on landfill operators. The first charge, imposed on all landfill operators, generates revenue for the municipality where the landfill is located to finance environmental protection activities. The second charge is imposed only on those landfills that do not adhere to specified waste disposal standards. Evidence suggests that the tax has markedly increased the proportion of sites attaining specified standards. One report indicates that the charge 'very positively motivated the establishment of new dumps in accordance with the strict required criteria concerning the safe storing of waste'. The amounts of both charges vary significantly according to the type of waste, the highest being 5000 Kc ($184) per metric ton for dangerous waste. A similar system operates in the Slovak Republic. It is more common for charges to be placed on generators of waste (rather than disposers of it), with applications in China, Estonia, Hungary, Poland and Russia.

In much of Eastern Europe and the former Soviet Union, charges on waste as well as air and water pollution are higher for quantities in excess of permitted levels or for improperly handled quantities. These higher incremental rates for levels in excess of standards could be looked upon as non-compliance fees.

Product charges

Levied in numerous industrialised countries, product charges are imposed either on a product or some characteristic of that product. Although some of these charges may discourage consumption, many of them are advance disposal fees intended to finance the proper disposal of the products after their use. Products on which charges have been imposed include automotive air conditioners (Canada), batteries (Canada, Denmark, Portugal and Sweden), beverage containers (Belgium, Finland, Norway and Sweden), building materials (Denmark), CFCs (Australia and Denmark), dry cleaning solvents (Denmark), fertilisers (Austria, Finland, Norway and Sweden), light bulbs (Denmark and Korea), lubricating oil (Finland, France, Italy, Norway and Spain), packaging (Belgium and Germany), pesticides (Belgium, Denmark, Norway and Sweden), plastic and paper bags (Italy, Iceland and Denmark), sulfur in oil (Finland, Norway and Sweden), and tyres (Taiwan and Canada). South Korea in 1993 imposed advance disposal fees on several products that are difficult to treat or recycle, but the rates are rather low.

Summary

Bargaining processes and efficient resource allocations

- Bargaining processes might bring about efficient resource allocations (and so might lead to the attainment of efficient pollution outcomes without regulatory intervention).
- The likelihood of efficient bargaining solutions to pollution problems being achieved is reduced by the presence of bargaining costs.
- Where the good in question is a public good, it is generally accepted that bargaining is impossible.

The instruments available to attain a pollution target

- Pollution control instruments can be classified into a set of broad classes, the most important of which are command and control instruments and economic incentive-based instruments.
- Our discussion of the properties and relative advantages of various instruments that could be used to attain environmental policy targets has taken place under the implicit assumption that some single authority has the ability to implement and administer a control programme. But many pollution problems spill over national boundaries. Given that the world does not have a single government, how can policy targets and instruments be devised, introduced, administered and monitored for global or international pollution problems? This question warrants separate attention in Chapter 9.

The mechanisms by which pollution instruments operate in attaining targets

- The long-run effects of pollution control instruments can be very different from their short-run effects, because of net income effects and impacts on the rate and direction of technological change.

The comparative merits of alternative instruments

- An instrument that attains a pollution target at least cost is known as a cost-effective instrument.
- A least-cost control regime implies that the marginal cost of abatement is equalised over all firms undertaking pollution control.
- In many – but not all – circumstances, economic incentive-based instruments are more cost-effective than command and control instruments.

The significance, in instrument choice, of whether a pollutant is uniformly mixing

- Where a pollutant is not uniformly mixing, the relative advantages of incentive-based instruments are considerably reduced. Some forms of marketable permit systems appear to offer the best prospect of attaining ambient pollution targets cost-effectively.

Further reading

Good general accounts of environmental policy instruments are to be found in Stavins (1992).

Baumol and Oates (1988) is a classic source in the area of environmental regulation. The whole book is relevant but it is quite difficult and formal. The theoretical basis for a political economy of environmental regulation is investigated in Boyer and Laffont (1999). Tietenberg (1992, chapters 14 to 20) provides an extensive and primarily descriptive coverage of specific types of pollution and the control techniques applied to each. Other good general accounts of pollution control policy are to be found in Fisher (1981, chapter 12), which discusses the work of Ronald Coase and the roles of wealth and bargaining power, Common (1995), Hartwick and Olewiler (1986) and Goodstein (1995). Fisher and Rothkopf (1989) consider the justification for public policy in terms of market failure. A possibility, which we touch upon in the next chapter, is that public intervention itself generates substantial costs. These costs may be sufficiently large to prevent intervention delivering positive net benefits. This notion of 'government failure' is analysed in Weimer and Vining (1992). Laffont and Tirole (1993, 1996) discuss the innovation incentive effects of permits when number is limited.

There are several national and international agencies that produce periodic surveys of environmental protection instruments and their effectiveness. Among these are various parts of the United Nations Organisation, the European Union, the United States EPA and the OECD. An extensive listing can be found on the Chapter 7 Links web page. References that the reader may find useful include OECD (1995), which surveys the use of environmental taxes and other charges used for environmental protection in the OECD countries; Anderson *et al.* (1997), US experience with economic incentives instruments; OECD (1997b), evaluating economic instruments for environmental policy; OECD (1999) for a detailed account of instruments used – and their effectiveness – in OECD countries; EPA (1999), economic incentives for pollution control in the USA; EPA (2001), US experience with economic incentives; EEA (2001), which considers ways of improving official environmental reporting; and EEA (2000), an online survey of environmental taxes in the EU.

Pearce and Brisson (1993) discuss the use of command and control instruments in the UK. Böhm (1981) considers deposit refund systems. Helm (1993, 1998) discusses possible reform of environmental regulation in the UK. Smith (1998) investigates taxation of energy.

Portney (1990) analyses air pollution policy in the USA, and Portney (1989) assesses the US Clean Air Act. Crandall (1992) provides an interesting analysis of the relative inefficiency of a standards-based approach to fuel efficiency in the United States. Kolstad (1987) examines the inefficiency losses associated with using undifferentiated taxes or other charges when economic efficiency requires that charges be differentiated across sources. Krupnick's (1986) paper on nitrogen dioxide control in Baltimore, discussed in the chapter, repays reading in the original.

Dales (1968) is the paper generally credited with having established the notion that marketable permits may be used for pollution control, and Montgomery (1972) derived the efficiency properties of marketable permits. For accounts of the use of market-based pollution control instruments see Hahn (1984, 1989), Hahn and Hester (1989a, 1989b), Opschoor and Vos (1989) and Tietenberg (1990, 1992). Jorgensen and Wilcoxen (1990a, b, c) analyse the impact of environmental regulation upon economic growth in the United States (but note that these papers are relatively difficult).

The following references deal with air pollution emissions trading programmes in developing countries: Ellerman (2001), SO_2 emissions in China; Blackman and Harrington (1999); Benkovic and Kruger (2001); Montero *et al.* (2000), Chile; and several papers in the *Journal of Economic Perspectives* (Summer 1998, Vol. 12, no 3). Some general accounts of air emissions problems and policies in India are found in Bose *et al.* (1997, 1998). Cowan (1998) considers the use of economic instruments for water pollution and abstraction. Gerard (2000) considers a variety of performance bonds (known as reclamation bonds) to analyse land reclamation after mining operations have ceased. Boyd (2001) reviews the need for financial assurance rules (also known as bonding requirements) to compensate for environmental damage that may arise in the future, given the prevalence of abandoned environmental obligations.

Enforcement issues and incentive compatibility (to be discussed in the next chapter) are analysed in Heyes (1998) and Laplante and Rilstone (1996). For a detailed analysis of issues concerning compensation in connection with distribution effects of tax changes, see Hartwick and Olewiler (1986, chapter 12), who also analyse the consequences of subsidies and taxes in the short run and the long run. The role and importance of non-convexities are discussed in Fisher (1981, p. 177), Portes (1970) and Baumol and Oates (1988). Second-best aspects of taxation, and possible double dividends from environmental policy, are discussed in Cremer and Gahvani (2001) and Parry *et al*. (1999).

The seminal text on non-point pollution is Russell and Shogren (1993). Others on this topic include Dosi and Tomasi (1994), Braden and Segerson (1993), Laffont (1994), Millock *et al*. (1997), Romstad *et al*. (1997), Segerson (1988) and Shogren (1993). For water pollution see Segerson (1990) and Ribaudo *et al*. (1999), and for non-point pollution from agriculture Vatn *et al*. (1997).

Useful accounts of instruments used in fisheries management include OECD (1997a) and the regular OECD publication *Review of Fisheries*, which covers changes in fishery management systems. Discussion of the idea of a safe minimum standard of conservation can be found in Bishop (1978) and Randall and Farmer (1995). The 'Blueprint' series (see, for example, Pearce, 1991a) provides a clear and simple account of the new environmental economics policy stance, in a rather ideological style. Finally, a number of texts provide collections of papers, several of which are relevant to pollution control policy: these include Bromley (1995) and, at a more rigorous level, the three 'Handbooks' edited by Kneese and Sweeney (1985a, 1985b, 1993).

Discussion questions

1. The text observed that strict liability and negligence liability both produce an outcome in which the socially efficient level of precaution is undertaken, but that their distributional impacts are different. In the light of the results given in the text about those distributional outcomes, what do you consider to be the relative merits of strict versus negligence liability?
2. Consider the case where the damages from polluting firms take the form of a public good (or public goods). Is there a case for the EPA to charge the injuring parties for the damage caused by the pollution or other kind of risky activity? And if the EPA were to obtain such damage payments, should the EPA distribute the moneys recovered from such damage settlements to the pollution victims?
3. Consider a good whose production generates pollution damage. In what way will the effects of a tax on the output of the good differ from that of a tax on the pollutant emissions themselves? Which of the two is likely to be economically efficient? (Hint: think about substitution effects on the demand side and on the supply side.)
4. Evaluate the arguments for the use of market or incentive-based instruments versus 'command and control' instruments in the regulation of environmental externalities under conditions of certainty.
5. Discuss the scope for the allocation of private property rights to bring the privately and socially optimal levels of soil pollution into line.
6. Discuss the distributional implications of different possible methods by which marketable permits may be initially allocated.

7. Suppose that marketable emissions permits are sold by competitive auction. Suppose also that an environmental protection interest group or NGO is seeking to purchase some emissions permits and that, if successful, it would retire them, thus reducing the total quantity of permits in existence. Should the EPA allow such bids?

8. Distinguish between private and public goods externalities. Discuss the likelihood of bargaining leading to an efficient allocation of resources in each case.

9. Use diagrams to contrast pollution tax instruments with marketable emission permit systems, paying particular attention to the distributional consequences of the two forms of instrument. (Assume a given, target level of pollution abatement, and that permits are initially distributed through sale in a competitive market.)

10. Discuss the efficiency properties of a pollution tax where the tax revenues are earmarked in advance for the provision of subsidies for the installation of pollution abatement equipment.

11. Suppose that a municipal authority hires a firm to collect and dispose of household waste. The firm is paid a variable fee, proportional to the quantity of waste it collects, and is charged a fee per unit of waste disposed at a municipal waste landfill site. Households are not charged a variable fee for the amount of waste they leave for collection, instead they pay an annual fixed charge. Comment on the economic efficiency of these arrangements and suggest how efficiency gains might be obtained.

12. An interesting example of a regulatory failure relates to electricity generating stations in the UK. Several thermal power stations in the UK were required to install flue-gas desulphurisation (FGD) plant in order to meet a national standard for sulphur emissions. The power stations fitted with FGD plant are not compensated for sulphur abatement. Electricity is purchased for the national grid on a competitive bidding system. The stations fitted with FGD are unable to compete on cost with other stations without that equipment, and as a result are withdrawn entirely from the grid at some times and operate below capacity at others.

    Explain why this situation is socially inefficient, and suggest a means by which this inefficiency could be avoided.

13. In Section 6.4.2.4, dealing with flexible permit systems with offsets, we explained the nature and role of offsets in marketable permit systems. But what is the motivation for offsets here? Why does the EPA not just extend the controlled group so as to get the benefits of cheaper abatement outside the group?

Problems

1. Suppose that an EPA must select one instrument from two available. Two criteria matter: (a) P, the probability of the instrument attaining its target; (b) C, the proportionate saving in abatement cost incurred in using that instrument (relative to the cost using the highest-cost instrument). The EPA calculates a weighted sum (score) for each instrument, and chooses that with the highest score. Assume that the instruments have the following values for P and C:

    Instrument 1: $P = 0.9$, $C = 0.0$

    Instrument 2: $P = 0.7$, $C = 0.2$

    (i) Write an Excel spreadsheet to illustrate how the instrument choice varies with changes in the relative weights (between zero and one) attached to the two criteria. Also explore how instrument choice varies as the magnitudes of P and C for each instrument vary.

(ii) Use an algebraic formulation of this problem to obtain expressions that allow these results to be shown analytically.

2. Using the Excel workbook *Leastcost.xls*, demonstrate that the cost penalty from sharing abatement equally between the two firms rather than using the least-cost distribution of abatement is larger the greater is the difference in the firms' abatement cost functions (as measured by the value of the slope parameter in the abatement cost functions).

3. The Coase theorem claims that a unique and efficient allocation of resources would follow from rational bargaining, irrespective of how property rights were initially allocated. Demonstrate that the distribution of net gains between bargaining parties will, in general, depend upon the initial distribution of property rights.

4. Show that a pollution tax on emissions and a subsidy to output producers for each unit of pollution reduction would, if the rates of subsidy were identical to the pollution tax rate, lead to identical outcomes in terms of the levels of output and pollution for a given sized industry. Explain why the distribution of gains and losses will usually differ, and why the long-run level of pollution abatement may differ when the industry size may change.

5. In all discussions of pollution abatement costs in this chapter, the fixed costs of pollution abatement were implicitly taken to be zero. Do any conclusions change if fixed costs are non-zero?

6. Demonstrate that in the simple special case of a uniformly mixing flow pollutant, in which the value of the damage created by the emission is independent of the location of the emission source or the time of the emission, the tax rate should be uniform over all polluters for the tax to be an efficient instrument (that is, it will be applied at the same rate per unit of pollution on all units of the pollutant).

7. Our discussion in this chapter has shown that if the control authority does not know the marginal damage function, it will not be able to identify the economically efficient level of pollution abatement, nor the efficient tax or subsidy level. Demonstrate that
   (a) knowledge of the pollution abatement schedule alone means that it can calculate the required rate of tax to achieve any target level it wishes;
   (b) if it knew neither the marginal damage nor the marginal abatement cost schedules, then it could arbitrarily set a tax rate, confident in the knowledge that whatever level of abatement this would generate would be attained at minimum feasible cost.

8. You are given the following information:
   (a) A programme of air pollution control would reduce deaths from cancer from 1 in 8000 to 1 in 10 000 of the population.
   (b) The cost of the programme is expected to lie in the interval £2 billion (£2000 million) to £3 billion annually.
   (c) The size of the relevant population is 50 million persons.
   (d) The 'statistical value' of a human life is agreed to lie in the interval £300 000 to £5 million.

   If the only benefit from the programme is the reduced risk of death from cancer, can the adoption of the programme be justified using an economic efficiency criterion?

9. In controlling emissions, there is an important difference between a command and control instrument and a tax instrument. Both require that the polluter pay the cost of attaining the emission reduction target. However, the tax instrument imposes an additional charge (on the emissions which remain at the target level of pollutions); this is not paid under a command and control regime. The failure to incorporate damage costs into the price of the product can generate distortions or inefficiencies in the economy. Kolstad (2000), from which this problem is drawn, gives an example in the paper-manufacturing industry. Suppose that paper can be produced using pulp either from recycled paper (which is non-polluting) or from virgin timber (which is

polluting). Compare the operation of a CAC instrument with a tax instrument applied to the manufacture of pulp from virgin timber, and show how this distorts (creates an inefficiency) in paper production.

10. This exercise involves using an Excel file to undertake some simulations regarding the relative costs of alternative instruments, and to interpret and comment on your results. Instructions for the exercise are given in *Pollution2.doc*; the Excel file is *Pollution2.xls*. Both of these can be found in the *Additional Materials* for Chapter 6.

11. Demonstrate that the negligence liability principle creates an incentive on firms to undertake the socially efficient level of precautionary damage reducing activity.

    Hint: What we are trying to do here is to show that negligence liability induces $Q = Q^*$. To demonstrate that this is so, one could ask which level of precaution minimises the sum of costs incurred in reducing risk and expected compensation payments. Up to $Q = Q^*$, expected compensation payments are indicated by the function shown in Figure 6.2.

CHAPTER 7 Pollution policy with imperfect information

It surely goes without saying that the real world of resource and environmental economics is an uncertain one.

Conrad and Clark (1987), p. 176

Learning objectives

Having read this chapter, the reader should be able to

- distinguish between uncertainty about pollution abatement costs and pollution damages
- understand how efficiency losses arise from making decisions under conditions of uncertainty
- appreciate why all types of pollution control instrument will, in general, involve efficiency losses under uncertainty
- analyse how the choice of pollution control instrument might depend on the relative slopes of control cost and damage functions, and so discuss the comparative merits of alternative instruments
- appreciate some of the implications of non-linearity or threshold effects in emissions damage functions for pollution control programmes
- recognise the conceptual difference between an efficiency loss arising from pursuit of an inefficient target, and an inefficiency loss from not achieving pollution reductions at least cost
- understand some consequences of asymmetry of information between the EPA, as regulator, and the regulated parties
- explain how, at least in principle, an EPA may elicit private ('inside') information about emissions abatement costs from regulated businesses
- understand the idea of the precautionary principle, and how it might be applied in the case of pollution control policy

Introduction

An environmental protection agency (EPA) will, in practice, find itself in the position of having to make choices with only limited information. In this chapter we bring together a set of issues that are all relevant to making choices about pollution control under conditions of imperfect information.

The words *risk* and *uncertainty* are often used to characterise various situations in which less than complete information is available. Risk is usually taken to mean situations in which some chance process is taking place in which the set of possible outcomes is known and probabilities can be attached to each possible outcome. However, it is not known which possible outcome will occur. Alternatively, all that may be known is what could occur (but not probabilities), a situation often described as uncertainty. A more extreme case – sometimes called radical uncertainty – concerns circumstances in which it would not be possible even to enumerate all the possible outcomes.

It will not be necessary in this chapter to differentiate sharply between these (and other possible) types of limited information. We shall often refer to them all by the generic term 'uncertainty'. Later in the text, in Chapter 13, it will be necessary to be more precise in the language used, and so more complete definitions will be given for the various forms of uncertainty.

In this chapter we shall be concerned with *choices* that the EPA has to make. Those choices may concern what kinds of pollution to control and by how much to reduce pollution (pollution targets) or they may be about how to achieve those goals (pollution instruments). In particular, it is the consequences of making those choices under conditions of uncertainty that is of central focus here. Our presentation follows the sequence of topics in the previous two chapters: Sections 7.1 and 7.2 largely concern target choices, while the following section is mainly about instrument choices. However, as you will see, the independence of targets and instruments – where it is legitimate to first establish a target *without* regard to the instrument or instruments available for its implementation and *then* select an instrument to attain that target – is difficult to maintain where information is imperfect.

More specifically, Section 7.1 looks at the difficulties faced by an EPA in setting standards exclusively on the basis of economic efficiency in a world of imperfect information and uncertainty. Partly in response to this, but also because of concepts developed in other disciplines, we then (in Section 7.2) discuss the use of some form of precautionary principle as a basis for pollution control policy. The importance of the precautionary principle is such that we shall not complete our analysis of it in this chapter; we shall return to consider this matter further at several points later in the book, particularly in Chapters 13 and 17.

Section 7.3 will have something to say about each of the following:

- the choice of pollution instrument where there is uncertainty about either pollution abatement costs or pollution damages (or both);
- implications of non-linearity or threshold effects in emissions damage functions;
- asymmetric information: where distinct groups of actors (here the EPA, as regulator, and the various regulated parties) have different sets of information available to them;
- how the EPA may improve the flows of information available to it, and in particular how it may elicit inside information about emissions abatement costs from regulated businesses.

Finally, Section 7.4 investigates the causes and consequences of regulatory failure. This is a situation in which public policy that is intended to bring about efficiency gains or achieve other stated objectives either fails to realise those gains to any large extent or, in extreme cases, leads to outcomes which are worse than the pre-regulation state of affairs.

7.1 Difficulties in identifying pollution targets in the context of limited information and uncertainty

Much of the discussion of efficiency-based pollution targets in Chapter 5 implicitly assumed that the policy maker was well informed, and so either knew – or by an investment of resources could discover – the relevant cost and benefit functions. Is this assumption reasonable? To address this question, it is useful to recall what the policy maker needs to know (or have reliable estimates of) in order to identify economically efficient emission or pollution targets.

The environmental agency must know the functional forms, and parameter values, of all the relevant functions for the pollution problem being considered. In particular, knowledge of the benefits and damages of pollution (or the costs and benefits of pollution abatement) is required. Moreover, as we showed in our treatment of convexity, it is not sufficient to know the values of such things near the current position of the economy; they have to be known across the whole range of possibilities.

Further, while it is pedagogically convenient to write about 'pollution' as if it were a single, homogeneous thing, it is clear that 'pollution problems' come in many distinct forms. Even for one type of pollutant, we have seen (in Chapter 5) that stock effects and spatial considerations imply that the appropriate functions vary from place to place and from time to time. Clearly, knowledge is required about a large number of functions, and it is not evident that knowledge about one case can simply be transferred to other cases.

Where does information about marginal costs and benefits of pollution abatement come from? In a world of certainty, identification of marginal costs and benefits would be a straightforward matter of the EPA collating and processing known information. Unless avoidable mistakes are made, the resulting magnitudes are known without error. But where information available to the regulator is imperfect (which is the normal state of affairs) estimation becomes necessary. Estimation will often involve the use of sampling methods and statistical inference from the sample data. But any such process necessarily involves sampling error, typically summarised in standard errors of the estimates, or associated confidence intervals.

These kinds of 'statistical' uncertainties could be avoided by using census rather than sample survey data; that is, by collecting information from the whole relevant population rather than from a part of it. However, the costs involved in acquiring, collating, validating and processing census-based information are likely to be prohibitive, and it will not generally be efficient to use census data.

Sampling error is not the only cause of uncertainty in obtaining information about the marginal costs and benefits of pollution abatement. Of more importance is likely to be the fact that the data collected may not properly represent what the investigator is seeking to obtain. There are many reasons for this, and we mention just a few of them here.

Consider first the estimation of abatement costs in a market economy. Relevant information about those costs is decentralised, and those who possess it may have incentives not to truthfully reveal it.[1] We discuss these incentives, and ways in which it might be possible to design information-collection mechanisms so that it will be in the interest of agents to reveal without bias the information that the regulator seeks, later in the chapter.[2]

Difficulties in indentifying the benefits of pollution abatement are more profound. Such benefits consist of avoided damages. Identifying abatement benefits typically involves a two-step process. First, the impacts of abatement have to be established. Second, monetary valuations are put on those impacts. But it is necessarily the case that these processes are, once again, done in the context of imperfect information. This is most evident in the first stage. Scientific knowledge about pollution impacts is far from complete, and arguably can never be complete because of the stochastic and complex nature of ecosystem functioning. Hence we cannot be sure about the biological, ecological, health, physical and other impacts of a pollutant.

Moreover, knowledge of those impacts is not sufficient for an EPA to establish efficiency-based targets: these 'physical' units need to be given monetary values in order that they can be compared with abatement costs. Two chapters of this book are devoted to the issue of valuation of environmental impacts under conditions of certainty (Chapter 12) and uncertainty (Chapter 13), and so we shall not pre-empt those discussions here. Suffice it to say here that, as with cost estimation from sample data, the available evaluation procedures yield statistical estimates, with point estimates being intrinsically uncertain. But more importantly, as we show in Chapters 12 and 13, valuation of environmental services is beset by a host of theoretical and practical problems, and there is little consensus about the validity of current valuation techniques.

Three further complications beset policy makers. First, the relevant costs and prices (on both benefit and cost estimation sides) needed for evaluation should be those that correspond to a socially efficient outcome; these may bear little relation to observed costs and prices where the economy is a long way from that optimum. Second, difficulties are compounded by 'second-best' considerations. If the economy suffers from other forms of market failure too, then the 'first-best' outcomes we investigated earlier are not efficient. Finally, limited information and uncertainty do not simply mean that decisions should be taken in the same way (but have less 'accuracy') as under conditions of full information. As we show in Chapter 13, there can be profound implications for appropriate decision making under conditions of risk or uncertainty.

[1] This, of course, is one of the reasons why command and control instruments are likely to not be cost-effective in practice, as distorted or biased information from decentralised agents means that the regulator cannot know each sources marginal abatement cost function.

[2] Our discussion has proceeded on the basis that it is the EPA itself that will itself collect the data and make inferences from those data. In practice that is often not the case. Such tasks are commonly outsourced, and/or the EPA will make use of estimated cost functions produced by academic or other specialised research institutions. But all such methods suffer from the problem that survey respondents may for strategic reasons reveal biased information when that information is likely to be available in the public domain.

7.2 Sustainability-based approaches to target setting and the precautionary principle

Even taking the perspective of an economist, one would be very reluctant to rely exclusively on efficiency-based targets given the difficulties identified in the last section. It seems sensible to at least give some weight to alternative approaches to pollution policy that explicitly address limited information and uncertainty.

Non-economists are generally suspicious of driving policy on what are perceived as narrowly economic grounds and are critical of the importance that is often attached to efficiency by economists in thinking about pollution targets. Natural scientists, environmentalists and ecologists typically regard stability and resilience – defined in the ways we outlined in Chapters 2 and 3 – as being more fundamental objectives. However, these objectives – on the one hand, population stability and/or ecosystem resilience, and on the other hand, maximisation of net economic benefits – are not necessarily mutually contradictory. Much of environmental economics (and, more so, ecological economics) consists of an attempted synthesis of the two. There are many ways in which that synthesis might be pursued. Several are explained in Common and Stagl (2005).

Two directions that have been espoused as general guides, and which could be interpreted as being part of such an attempted synthesis, are to target policy at achieving sustainability, and to adopt some form of precautionary principle. In many respects, a sustainability approach is implied by the precautionary principle, and so we shall not address it separately here. In this section, we consider an approach to setting environmental targets or 'standards' which places less weight on economic efficiency and gives high weight to security and sustainability as policy objectives.

In a world of certainty (and so complete predictability) taking precautions would be unnecessary. But in a stochastic or complex environment where outcomes are not certain, where processes are incompletely understood, or where non-linearities of various kinds are thought to exist, some form of 'playing safe' is sensible. The precautionary principle – in some of its guises at least – can be thought of as a hybrid criterion. It tries to bring together efficiency, sustainability, ethical and ecological principles, into a bundle that can inform target setting. Of course, in trying to do several things at the same time, it runs the risk of not doing any of them particularly well. But the approach is now being widely advocated.

Suppose, for example, that unregulated pollution levels pose threats to the quality or availability of some natural resource (such as European marine fisheries or tropical forests) or jeopardise a more broadly defined environmental or ecological system (such as a wilderness area characterised by extensive biodiversity). In such circumstances, sustainability might be regarded as of greater importance than efficiency. Of course, if efficiency and sustainability criteria yielded identical policy recommendations, their relative importance would not matter. But as our analysis in Chapter 3 suggested, and as we demonstrate more thoroughly in Chapters 14 to 19, they do not. In general, the efficiency criterion is not sufficient to guarantee the survival of a renewable resource stock or environmental system in perpetuity.

The precautionary principle can be thought of as proposing a lexicographic approach to setting targets. We regard one criterion (in this case sustainability) as being of overriding significance, and require that any target do as well as possible in terms of this measure. If this leaves us with more than one option, then other desirable criteria can be employed (perhaps in a hierarchical way too) to choose among this restricted choice set. Alternatively, a constraint approach could be adopted: pollution policy should in general be determined using an efficiency criterion, but subject to an overriding sustainability constraint.[3]

Examples of the latter kind are given in Chapter 13, where we explain the notion of a safe minimum standard (SMS) of conservation, and in Chapter 17. When applied to pollution policy, the adoption of an SMS approach entails that threats to survival of

[3] The difference is that the lexicographic approach entails maximising objectives sequentially, whereas constraints only need to be satisfied.

valuable resource systems from pollution flows are eliminated. This is a strict interpretation of SMS. A modified SMS would eliminate the pollution flow, provided that so doing does not entail 'excessive cost'. It remains, of course, to determine what an 'excessive cost' is. This formulation of pollution policy recognises the importance of economic efficiency but accords it a lower priority than conservation when the two conflict, provided that the opportunity costs of conservation are not excessive. This compromise between efficiency and conservation criteria implies that 'correct' levels of pollution cannot be worked out analytically. Instead, judgements will need to be made about what is reasonable uncertainty, what constitutes excessive costs, and which resources are deemed sufficiently valuable for application of the SMS criterion.

SMS is one example of a wider set of concepts that all embody some form of precautionary principle. Most statements of the precautionary principle begin with an explicit recognition of the presence of uncertainty. Where environmental policy choices have to be made, we do not and cannot have full information. Outcomes of choices cannot be known with certainty. Given that possible outcomes may include ones which are catastrophic, in such circumstances the policy maker may choose to play safe, adopting a presumption about not changing conditions too much from the *status quo ex ante*. This is explored at more length in Chapter 13.

7.3 The relative merits of pollution control instruments under conditions of uncertainty

Table 6.1 listed a set of criteria that could be used to appraise the relative advantages of alternative types of pollution control instruments. Five of these – cost-effectiveness, long-run effects, dynamic efficiency, ancillary benefits, and equity – were discussed in Chapter 6. We listed, but did not discuss, four others, noting that they all relate in some way to the issue of instrument choice under conditions of uncertainty. Before we take up this thread again, note that one particular kind of uncertainty did play an important role in the conclusions reached in Chapter 6. There, we recognised that the EPA is unlikely to know the marginal abatement cost functions of individual firms. Indeed, it was precisely this which underpinned the contention that economic incentive-based instruments have an important advantage over command and control regulations. Incentive-based instruments are often able to attain targets at least cost even where the regulator has no information about individual firms' abatement costs. This is not true when the EPA uses a command and control technique. Important as that matter may be, it only deals with one, relatively narrow, facet of uncertainty. It is now time to broaden the discussion. To do so, we bring into consideration the four criteria listed in Table 6.1 that were not examined earlier. For convenience, these are restated in Table 7.1. We now examine each of them in turn.

Table 7.1 Additional criteria for selection of pollution control instruments

Criterion	Brief description
Dependability	To what extent can the instrument be relied upon to achieve the target?
Flexibility	Is the instrument capable of being adapted quickly and cheaply as new information arises, as conditions change, or as targets are altered?
Costs of use under uncertainty	How large are the efficiency losses when the instrument is used with incorrect information?
Information requirements	How much information does the instrument require that the control authority possess, and what are the costs of acquiring it?

7.3.1 Dependability of the control instrument

An instrument is dependable if it can be relied upon to achieve a predetermined target. Where the EPA has full information, each of the instruments discussed in the previous chapter has this property. Emissions quantity controls – whether or not they are marketable – can be set directly to the targeted emissions. Price controls (emissions taxes and abatement subsidies) will also have known quantity outcomes, and so can be set at whatever level is necessary to achieve the relevant objective. For example, with full knowledge of the aggregate

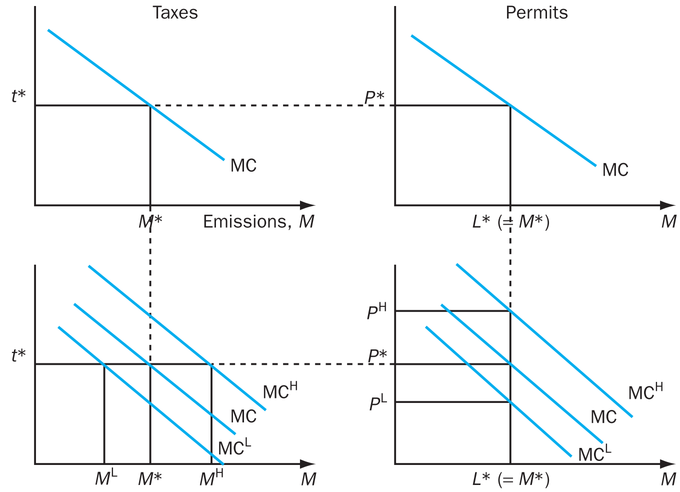


Figure 7.1 A comparison of emissions taxes and marketable permissions permits when abatement costs are uncertain

abatement cost function, the EPA can determine what emissions tax rate (or emissions abatement subsidy) is needed to achieve any given level of abatement. Once that tax rate is introduced, abatement takes place at the desired level, as shown in the top half of Figure 7.1. Moreover, the aggregate target would be achieved at least cost.

But now consider one particular form of imperfect information: the EPA does not know the location of the aggregate emission abatement cost function with certainty. Then price-based instruments (taxes and subsidies) and quantity-based instruments (licences and marketable permits) differ. With an emissions tax the amount of abatement that results from any given rate of tax will not be certainly known, as it will depend upon the unknown position of the abatement cost function. Licences and marketable permits are dependable in terms of abatement (although there will be uncertainty about the size of abatement costs and, with marketability, the price of emissions permits).

The differing way in which abatement cost uncertainty affects taxes and marketable permits is illustrated in Figure 7.1. In the upper half of the figure, a single aggregate marginal abatement cost function (MC) is drawn, assumed to be known by the authority.[4] Tax and permit regimes are identical in outcomes. In contrast, abatement cost uncertainty is represented in the lower half of the diagram by showing three different realisations of marginal abatement costs. These three curves can be thought of as three drawings from a probability distribution that describes the whole set of possible outcomes. It is evident that quantity-based controls can have very different impacts from price-based controls. In general, uncertainty about abatement costs translates into uncertainty about the quantity of abatement in a price system (such as emissions taxes or emissions abatement subsidies). It translates into uncertainties about prices or costs under a quantity-control system. For example, the aggregate marginal abatement cost and the marginal abatement costs for individual firms will be uncertain under a non-tradable permits system, or the equilibrium permit price, *P*, and aggregate marginal abatement cost will be uncertain under a marketable permits system.

It is sometimes claimed that command and control instruments – or more specifically emissions quotas

[4] As in previous chapters, we can think of a marginal abatement cost function as the profits forgone from various levels of emissions abatement. There is an opportunity cost to firms if they are required, or induced, to abate emissions, hence the use of our 'MC' terminology. Note that abatement becomes greater (that is, emissions are reduced) as we move from right to left in the diagrams used in this chapter.

and non-tradable licences – allow the EPA to control pollution outcomes more dependably than other instruments in situations of uncertainty. However, the preceding comments suggest that they do not have that property to any greater extent than tradable permits.[5] Moreover, some command and control instruments are clearly not dependable. The emissions outcomes from using technology requirements, for example, cannot be known *a priori*. It is evident that only emission quotas/licences or tradable permit systems can achieve emissions targets dependably in conditions of uncertainty.

7.3.2 Flexibility

Where decisions are made with perfect information, flexibility of an instrument is of little or no value. But with uncertainty, the EPA might need to learn adaptively by trial-and-error methods, and to change the rates or levels at which instruments are applied as new information arrives. Other things being equal, the more flexible an instrument is the better.

It is difficult to draw general conclusions about the relative flexibility of different instruments, as much depends on the specific circumstances that prevail. The literature contains a number of observations, some of which we outline in this section. However, their validity, or practical usefulness, remains unclear. One common assertion is that price-based incentive instruments (that is, tax or subsidy schemes) are inflexible as there is inherently strong resistance to changes in tax or subsidy rates. Changes in licences or permits do not seem to engender such strong resistance.[6] Command and control regulation might also be more flexible relative to schemes that are theoretically more attractive in economic terms (such as the ambient permit system described in Chapter 6). The latter schemes may be difficult and costly to design and, once established, will not easily admit piecemeal changes that could be done using command and control. Whether or not these assertions are empirically valid is, however, a moot point.

Another way of thinking about flexibility is in terms of whether price changes or quantity changes, brought about by new information, are socially less desirable. If the EPA judges that quantity variations are worse than price changes (perhaps because the former incur much larger transactions or adjustment costs), then instruments directed at quantities have an advantage.

Technology regulations can be inflexible in a particular way. They direct producers to do things in certain ways, and so impose large capital costs that are regulation-specific. Changes in those regulations lead to (potentially large) re-equipping or re-engineering investments. So technology requirements may be inadequate where new information is continually being obtained. Finally, we note that flexibility of an instrument is related to the transactions costs associated with its use. The greater are these costs, the less will the EPA wish to change the instrument setting. Transactions costs are discussed in Section 7.4.

7.3.3 Costs associated with uncertainty

Making choices in circumstances where everything is known or perfectly predictable is fundamentally different from making choices where that is not the case. In the former situation mistakes are avoidable, and are generated only by computational error or by limits to computational capacity. Pollution abatement will, of course, involve real resource costs as resources have to be devoted to pollution control, or techniques have to be used that would not otherwise be selected. These costs make up the abatement cost functions we have been referring to throughout this book. The regulator will also incur costs: those associated with processing information, and implementing, monitoring and enforcing the pollution control programmes.

[5] Licences may be more dependable than permits for a non-UMP pollutant where location matters, as the consequences of trading may then be unpredictable in terms of pollution concentrations.

[6] This perception may of course be erroneous. As tax or subsidy changes affect all members of a particular class of polluters, opposition is likely to be collective and deliberately made public, with warnings of potential job losses or the like used in attempts to harness public support or legitimacy for resistance to change. Changes in licences, particularly of the command and control non-tradable variety, affect individual polluters differentially, and so are less likely to elicit collective and public responses. Nevertheless, one would expect that affected units will lobby very hard to avoid tightening of restrictions and, to the extent that regulatory capture exists, may be successful in their efforts in an unobserved way.

However, in an uncertain world, decisions may have to be made before all information that is relevant to that choice is known. Not only may computational mistakes be made, but also a second class of 'error' can be made. Choices made today using available information will sometimes turn out – with the benefit of hindsight – to be less good than some other choice would have been.[7] This kind of error is unavoidable in situations of uncertainty, and is conceptually distinct from avoidable 'mistakes'. Nevertheless such errors will generate costs that are additional to those already described. Two of these additional 'uncertainty' costs are relevant to our present discussion:

1. costs incurred as a result of the selection of incorrect targets;
2. costs incurred by failing to attain aggregate targets at least cost.

For instrument choice, it is important to have some idea about how large the costs associated with these errors might be using one instrument rather than another. If one instrument turned out to always carry greater cost-penalties from making these errors than any other instrument would, that would create a strong presumption against its use.

The costs associated with (unknowingly) selecting the wrong target are 'efficiency losses' or 'welfare losses'. It is important to be clear about what kind of loss we have in mind here. Figure 7.2 helps to fix ideas. This establishes a 'baseline' against which the efficiency losses from errors due to uncertainty can be measured. The efficient target, M^*, is that level of emissions which equates the marginal cost of emissions abatement (MC) and the marginal damage of emissions (MD). The shaded area in Figure 7.2 represents the total net social benefit that would be generated at that level of emissions. This is the maximum net benefit available. The efficiency losses we have in mind are those in which emissions are at any level other than M^*, and so attained net benefits fall short of their maximum level.

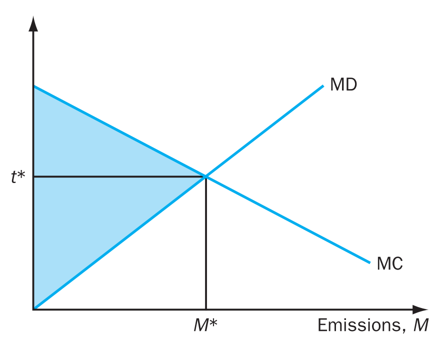


Figure 7.2 Target setting under perfect information

7.3.3.1 Uncertainty about abatement costs

Uncertainty about abatement costs may result in an efficiency loss of this kind. Suppose that the EPA knows the pollution marginal damage function (MD) but has to estimate the marginal emissions abatement cost function (MC), and will often make errors in doing so. Overestimation and underestimation of abatement costs will each lead the EPA to wrongly identify the efficient level of emissions, and so to an efficiency loss. But, as we shall see, the magnitude of that loss will differ depending on which instrument the EPA chooses to use. Let us investigate the relative magnitudes of efficiency loss under an emission tax system and an emission licence scheme.

Figure 7.3 shows the case in which the marginal cost of abatement is overestimated. Consider first an emissions fee. On the (incorrect) assumption that the marginal abatement cost curve is the one labelled 'MC (assumed)', the EPA imposes a tax at the rate t^H (as opposed to its true efficiency level, t^*). Firms will abate emissions as long as their actual (true) marginal abatement costs are below the tax, and so will emit at M^t, a rate less than the efficient level. The resulting efficiency loss is defined by the shortfall of net benefits at M^t compared with the maximum obtainable level at M^*; this is indicated by the heavily shaded area in the diagram.

Compare this efficiency loss with that which results from using an emissions licence system. Using incorrect information, the EPA believes the efficient target is L^H (when in fact it should be M^*). Incorrect information has led the regulator to pursue

[7] There are circumstances in which it will never be known whether the choices that have been made were the best ones (or at least not until an avoidable catastrophe takes place which reveals that the choices made were not wise).

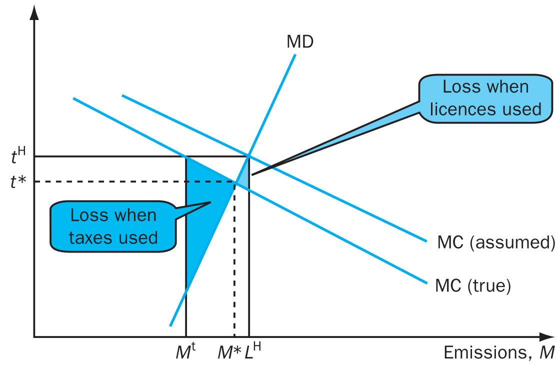


Figure 7.3 Uncertainty about abatement costs – costs overestimated

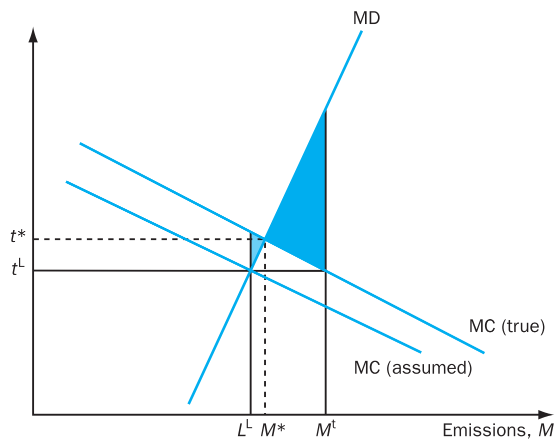


Figure 7.4 Uncertainty about abatement costs – costs underestimated

an insufficiently tight control. The efficiency loss is indicated by the lightly shaded area (corresponding to the surplus of marginal damages over marginal abatement costs for the excessive units of emissions).

Of course, errors may also take the form of underestimation of abatement costs. This is represented in Figure 7.4, in which the shapes and positions of the 'true' functions are identical to those in Figure 7.3 to allow direct comparison of the two diagrams. Now the assumed marginal abatement cost curve lies below its true position. Using similar reasoning to that given above, it can be seen that an emissions tax results in a loss (shown by the heavily shaded area) that is greater than the loss associated with licences (the lightly shaded triangle).

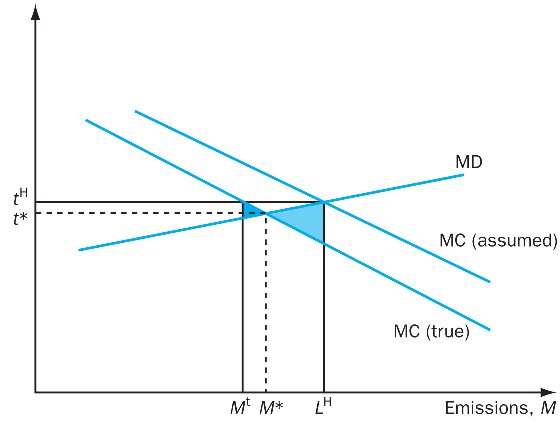


Figure 7.5 Uncertainty about abatement costs – costs overestimated

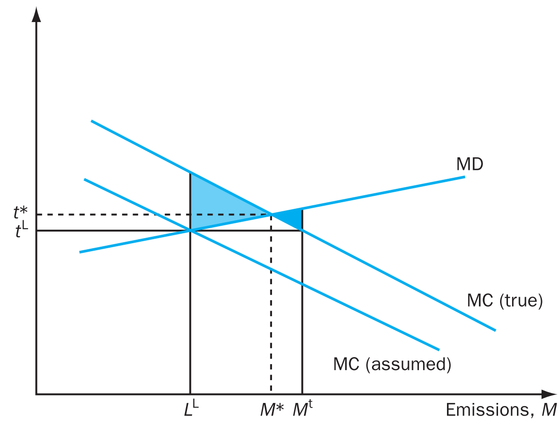


Figure 7.6 Uncertainty about abatement costs – costs underestimated

An incorrectly estimated abatement cost function results in an efficiency loss. In the case we have investigated, irrespective of whether the error is one of over- or underestimation, the loss from using taxes exceeds that from using licences. However, this result depends on the manner in which we constructed the functions in Figures 7.3 and 7.4. Compare these with the cases shown in Figures 7.5 and 7.6. These are analogues of the two situations just investigated, but are drawn with a substantially flatter marginal damage curve. Once again, both instruments generate efficiency losses when mistakes are made about abatement costs. But the ranking of the two instruments is now reversed: the loss is larger with licences than with a tax.

It turns out to be the case that what differentiates these two pairs of cases is the relative slopes of the MC and MD functions. We obtain the following general results:

- When the (absolute value of the) slope of the MC curve is less than the slope of the MD curve, licences are preferred to taxes (as they lead to smaller efficiency losses).
- When the (absolute value of the) slope of the MC curve is greater than the slope of the MD curve, taxes are preferred to licences (as they lead to smaller efficiency losses).

7.3.3.2 Uncertainty about pollution damages

The arguments so far have been conducted in the context of uncertainty about abatement costs. The conclusions we reached do not carry over to uncertainty about damage costs. In this case, the choice of quantity- or price-based instruments has no bearing on the magnitude of the efficiency loss arising from errors in estimating damage costs. The size of that loss is the same in each case, and so knowledge about the relative slopes of functions can give no information that would minimise such losses. This result is illustrated in Figure 7.7.

Given the estimated marginal damage function and the marginal cost function (assumed here to be correctly estimated), an EPA might set a tax at the rate *t* or a quantity control at the amount *L*. In either case, the level of realised emissions exceeds the efficient level M^* and the efficiency loss associated with the erroneous target is shown by the shaded area in Figure 7.7.

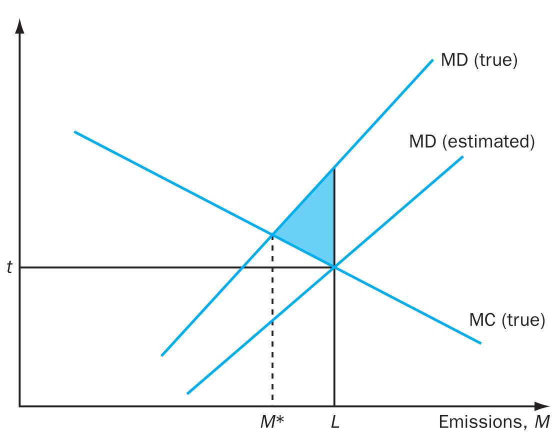


Figure 7.7 Uncertainty about damage costs – damages underestimated

The reason why errors in damage estimation and errors in abatement cost estimation have different implications concerns the behaviour of abating firms. Where errors are made on the damage side, a tax scheme and a quantity control are coincident in their effects on abatement. But where errors relate to abatement costs, the divergence between estimated costs (which determine the level of regulation imposed by the EPA) and true costs (which determine behaviour of polluting firms) drives a wedge between the realised emissions of firms under price- and quantity-based regulation.

It is important to note that the results we have derived so far relate only to a particular – and very limited – form of uncertainty, in which the general form of the damage function is known, but its position cannot be estimated with certainty. Where uncertainty about damages is of a more profound, radical form, an entirely different approach to setting targets (and choosing instruments) may be warranted, as we suggested in Section 7.2.

7.3.3.3 The consequences of a threshold effect in the pollution damage function

In this section we continue to assume that there is uncertainty about the location of the MC function, but now assume that the pollution damage function is known to contain a threshold effect. Can any conclusions be obtained about the best choice of instrument in the case? We go through an argument used in Hartwick and Olewiler (1998) that generates some interesting insights into this question. Hartwick and Olewiler present the situation shown in Figure 7.8. The total damage function contains two linear segments, with a discontinuity ('threshold') at emission level M^{∇}. This total damage function corresponds to the marginal damage function portrayed in panel (b) of the diagram. As with the total function, marginal damages again exhibit a discontinuity, although they are constant above and below that discontinuity (because of the linearity of the two segments of the total damage function).

The case we investigate is one in which the EPA knows the shape and position of the marginal

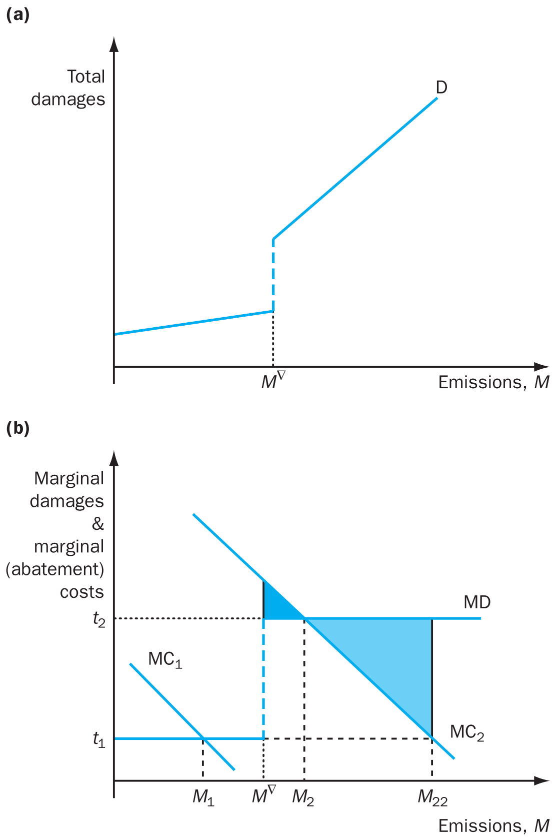


Figure 7.8 Consequences of a threshold in the damages function

damage function, and so is aware that a threshold exists at $M = M^\nabla$. However, as in Section 7.3.3.1, the EPA is uncertain of the location of the marginal abatement cost function. Two of the many possible locations of the marginal abatement cost curve are labelled as MC_1 and MC_2. Suppose that the EPA estimates that marginal costs are given by the curve MC_1, and sets an emissions tax at the rate t_1. If the EPA's estimate of MC were true, this would yield the efficient level of emissions, M_1. Even if that estimate were incorrect by a relatively small amount that tax rate would still generate an efficient level of emissions.[8] More precisely, provided that the true value of MC is such that its intersection with MD is somewhere between $M = 0$ and $M = M^\nabla$, the tax rate t_1 would induce an efficient emission. Note, in contrast, that a quantity control would not have this attractive property.

However, suppose that the EPA had grossly underestimated marginal abatement costs, with the true function actually being MC_2. Inspection of the diagram makes it clear that a tax rate set at the value t_1 would lead to substantially excessive emissions. Efficient emissions would be M_2 but realised emissions are M_{22} (with an efficiency loss shown by the lightly shaded area in Figure 7.8b).

If the tax rate had been set at t_2, efficient emissions result if the true value of MC lies in the neighbourhood of MC_2, but it imposes a massively deficient emissions outcome (here zero) if MC_1 is the actual value. Overall we see that the non-linearity in damages implies that a price-based policy has attractive properties where errors are not too large. However, when the estimation error goes just beyond some critical size, the efficiency loss can switch to a very large magnitude.

We leave the reader to explore the use of quantity controls. You should find that if the EPA set a control at the quantity M_1 or M_2 (depending on which MC function it deems to be relevant), the likelihood of extremely large efficiency losses is reduced, but at the expense of losing some efficiency for relatively small errors in estimation.

Hartwick and Olewiler conjecture that a best policy in the case analysed in this section is one that combines a tax (price) control and an emissions (quantity) control. They propose a tax equal to the lower value of the MD function, and an emissions limit equal to the threshold level. The tax bites – and generates efficient emissions – if marginal abatement cost lies in the neighbourhood of MC_1. Where MC is sufficiently large to intersect MD in its upper segment, the quantity constraint bites. Such a composite policy does not eliminate efficiency losses, but it prevents such losses being excessively large. Finally, the authors argue that such a combined policy is also prudent where uncertainty surrounds the position of the MD function. We leave analysis of this case as an exercise for the reader.

[8] This result arises from the fact that the MD curve is horizontal in this neighbourhood. If it were not, this efficiency property would not hold exactly.

7.3.3.4 General conclusions

Collecting the relevant results together, we can summarise as follows. Consider first the case where functions are linear, and uncertainty relates to the marginal abatement cost (MC) function. Then an EPA should prefer a quantity policy (licences) to an emissions tax if MC is flatter than MD, and an emissions tax to a licence system if the reverse is true, if it wishes to minimise the efficiency losses arising from incorrect information. However, where uncertainty pertains to the MD function, knowledge of relevant slopes does not contain information that is useful in this way. Once the existence of non-linearity and/or threshold effects is admitted, general results become much harder to find. In some circumstances at least, combined tax–quantity-control programmes may have attractive properties.

It is clear that the presence of uncertainty substantially weakens the general presumption in favour of incentive-based instruments over quantitative regulations that we developed in the previous chapter. They may be better in some circumstances but not in all. Finally, we note that experience with using taxes/subsidies or quantity controls will tend to reveal information through time that may help to reduce uncertainty. The logic behind this claim will be explained in the next section.

7.3.4 Information requirements: asymmetric information and incentive compatibility

The analysis so far in this chapter has shown that imperfect information puts restrictions on the ability of the EPA to devise 'good' targets and to attain them at least cost. It also considerably complicates its choice of instrument because comparative advantages depend on the prevailing circumstances. Moreover, limited information and uncertainty may prevent the EPA from knowing which circumstance actually pertains. Faced with all this, there are strong incentives on the EPA to become better informed. One would expect that it would invest in systems that deliver greater information. There are three ways that the EPA might do this:

- undertake its own research to gather data;
- build long-term institutional relationships with regulated businesses;
- create reward structures that give firms incentives to reveal information truthfully to the regulator.

There are limits to how far these can be taken. It is clear that undertaking (or commissioning another body to undertake) research is a costly exercise. Moreover, it is possible that, well before complete knowledge has been attained, the incremental costs of additional research activity will exceed its benefits. The second method – building institutional relationships with polluting businesses – has much to offer, particularly in terms of the prospect of access to private company-level information. But the approach also has drawbacks. Most importantly perhaps is the possibility of 'regulatory capture', in which the relationships threaten to undermine the independence of the regulator from the regulated parties.

Environmental economists have focused increasingly on the third of these options, looking for 'incentive-compatible instruments'. An instrument is *incentive-compatible* if the incentives faced by those to whom the instrument applies generate behaviour compatible with the objectives of the regulator. In general, none of the instruments we have discussed so far has this property. Where polluters think that the numbers they report can influence the severity of regulation, they have an incentive to lie about the costs of complying with abatement targets. This is true whether the instrument being used is command and control, emissions tax, abatement subsidy or a marketable permit scheme.

In the following section, we illustrate two examples of such incentive effects. If firms expect tax schemes to be used they have an incentive to understate abatement costs. If they expect a marketable permit scheme, the incentive is to overstate these costs. We also outline one possible instrument – a mixture of abatement subsidy and marketable permits – that is incentive-compatible.

7.3.4.1 The incentives to be untruthful under tax and marketable permit regimes

Case 1: Firms expect a permit system to be in operation

Suppose that firms expect the EPA to use a marketable permit system. Moreover, they believe that

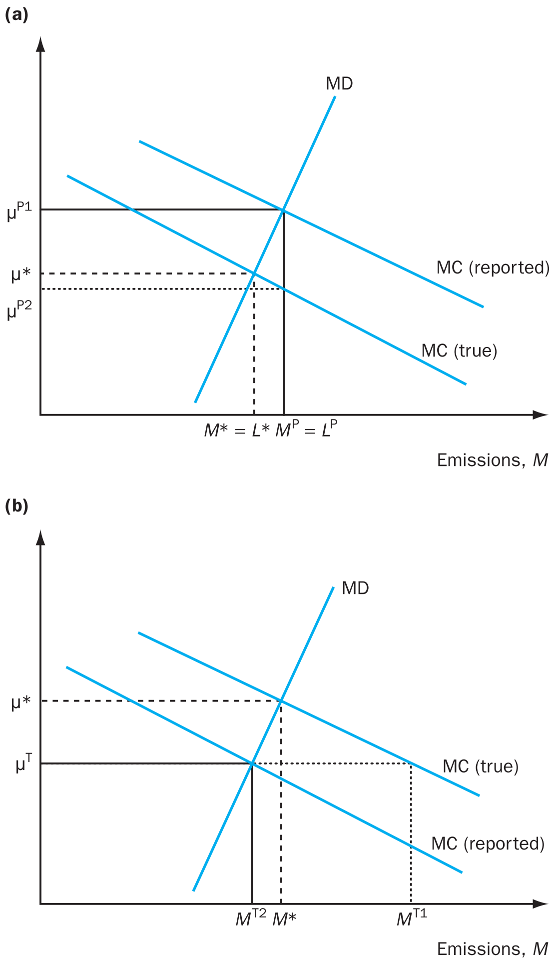


Figure 7.9 (a) Incentive effects with permit systems; (b) Incentive effects with an emissions tax

the total number of permits issued will be equal to what the EPA estimates to be the economically efficient level of emissions. Finally, firms realise that the EPA will make its choice of permit quantity only *after* firms have provided the EPA with information about their emissions abatement costs.

Using Figure 7.9(a), we will show that it is in the interest of firms to exaggerate their marginal abatement costs (MC).[9] If firms honestly report their actual abatement costs, L^* permits are issued, allowing an efficient emissions level M^*.[10] In a competitive permits market the equilibrium permit price would be μ^*. If firms lie, and overstate their abatement costs, the EPA incorrectly believes that the efficient target is M^P, and so issues that number of permits (L^P). Exaggerating abatement costs is better for firms than being truthful as more permits are issued, and so they incur lower real emission abatement costs.[11]

Note in passing a point that we will return to later, and which was alluded to earlier in remarking that the use of instruments will reveal useful information. The EPA expects the permit price to be μ^{P1}, although the actual price will turn out to be μ^{P2} (because the true marginal abatement cost function is the demand curve for permits).

Case 2: Firms expect an emissions tax to be in operation

Now we suppose that firms expect the EPA to use an emissions tax system. Equivalently to Case 1, firms expect that the EPA will set what it believes to be the efficient tax rate only after firms have informed the EPA of their abatement costs. Figure 7.9(b) shows that firms have an incentive to understate their abatement costs. If abatement costs are reported truthfully, a tax rate of μ^* is set, leading to emissions of M^* (the efficient level). However, if firms understate their abatement costs the EPA incorrectly believes that the efficient tax rate is μ^T, and sets that rate. This results in a quantity of emissions M^{T1}. Firms benefit because they emit more than if they told the truth (and so incur lower real abatement costs). Also, the tax rate is lower than it would be otherwise.[12]

Note also that the EPA expects the quantity of emissions to be M^{T2} whereas in this scenario it will

[9] The functions shown are aggregate (industry-wide) curves, not those for a single polluter.

[10] M^* is the target if an efficiency criterion is used by the EPA. But the arguments used in this section do not depend on targets being chosen in that way. Any upward-sloping function (replacing the marginal damage function) would generate the same results about incentives to lie.

[11] The size of the actual financial gain also depends on whether or not permits are initially allocated without charge. If they are, the only change to firms' financial position is the reduction in abatement costs. If the permits are purchased via auction, then there is an additional factor to take into account: the permit price is lower, but more permits are bought. Nevertheless, firms must still gain overall.

[12] The total tax bill might rise because more emissions take place; but firms must still gain overall, as one option they have available is to emit no more but pay the lower tax on those emissions.

turn out to be M^{T1}. Once again, this information will prove useful to the regulator. Indeed, whether a tax or a permit system is used, untruthful behaviour is revealed after the event to the EPA. The EPA observes a difference between what it expects the permit price to be and what it actually is (or between the actual and expected levels of emissions). Moreover, it will be able to deduce in which direction abatement costs have been misreported. So one possibility open to the EPA is to adopt an iterative process, changing the number of permits it issues (or tax rate) until there is no difference between actual and expected outcomes. But this may not be politically feasible, or it may involve large costs in making successive adjustments.

7.3.4.2 An incentive-compatible instrument

Can an instrument be found which will encourage truthful behaviour *and* allow the EPA to achieve its objective? More specifically, what we are looking for here is an instrument that creates an incentive to report abatement costs truthfully and which allows the EPA to achieve whatever target it wants in a cost-effective way. Several schemes with such properties have been identified. We examine one of them, proposed by Kwerel (1977).

The scheme involves a combination of marketable permits and subsidies on 'excess' emissions reduction. Some intuition can help to understand why this will work. The costs that firms report have two effects: they influence the number of permits issued, and they also influence the subsidy received for excess emissions reduction. The scheme balances these two influences so as to reward truthful reporting. Kwerel's scheme works in the following way. Firms are told that:

1. permits will be allocated through auction;
2. they will receive a subsidy for any emissions reduction over and above the number of permits they hold;
3. the subsidy rate will be set at the intersection of the marginal damage curve and the reported marginal abatement cost curve.

Given this information, firms are then asked to report their abatement costs, the subsidy is set accordingly, and the permit auction takes place.

The total cost of the scheme to all firms in the industry is equal to actual emission abatement cost plus the cost of acquiring permits less the subsidy payments received on any emissions reduction over and above the permitted amount of emissions. We use the following notation: L = number of permits made available to industry; P = price of permits; s = subsidy per unit of emissions reduction. Then we can write an expression for pollution abatement costs (PCC) for the whole industry.

$$\text{PCC} = \underset{\text{(area under MC curve)}}{\text{Abatement costs}} + \underset{P \times L}{\text{Permit costs}}$$

$$- \underset{s \times (L - M)}{\text{Emissions reduction subsidy}}$$

To demonstrate that this instrument is incentive-compatible, we compare the benefits to firms of being truthful with the benefits of (a) understating costs and (b) exaggerating costs.

Case 1: Firms understate abatement costs (see Figure 7.10(a))

Understating causes permits to be scarce ($\check{L}$ rather than L^*). The permit price is driven up to $\check{P}$, the level determined by true abatement costs. Hence total costs rise because (a) fewer emissions licences means it must do more abatement (heavily shaded area) and it has to pay a higher permit price (lightly shaded area). The combined area of these is larger the greater is the permit price. Note that the firms' total costs are increasing in the permit price. It follows from this that firms' costs are minimised when the costs reported are actual costs, which drives the permit price down to its lowest level.

Case 2: Firms exaggerate abatement costs (see Figure 7.10 (b))

At first sight, exaggeration of abatement costs seems to be advantageous to firms: it increases allowed emissions (to $\bar{L}$) and it increases the subsidy rate (to $\bar{s}$). But there is another factor that dominates these considerations. The existence of the subsidy, $\bar{s}$, puts a floor (a minimum level) on the permit price. That price cannot fall below $\bar{s}$. For if it began to do so, firms would buy permits in order to receive

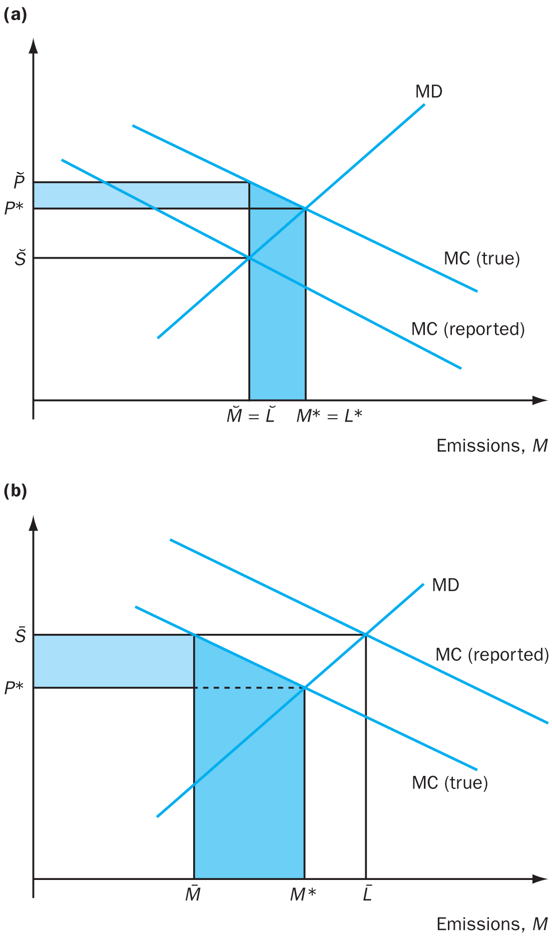


Figure 7.10 (a) An incentive-compatible instrument and under-reporting costs; (b) An incentive-compatible instrument and over-reporting costs

the (higher-valued) subsidy payment from holding more permits. But if the permit price is equal to $\bar{s}$, then the amount of permits actually bought will be $\bar{M}$ (even though a larger quantity $\bar{L}$ is available for purchase).

Figure 7.10(b) shows the losses that firms make as a result of exaggerating abatement costs. The heavily shaded area is the additional abatement costs, the lightly shaded area is the additional price paid for permits. It can be seen that as MC (reported) goes towards MC (true), these losses disappear. The best that the firm can do is to be truthful!

7.4 Transactions costs and environmental regulation

In carrying out its responsibilities, an environmental protection agency necessarily incurs transaction costs. This is a generic term for a variety of costs that include:

- acquiring relevant information;
- creating, monitoring and enforcing contracts (of which one category is the EPA's regulations);
- establishing, implementing and revising the instruments it employs;
- monitoring performance, and ensuring compliance.

It is important to be clear about what should and should not be included in the term 'transactions costs'. They *do include* the costs of the personnel and the structures an organisation puts in place that allow it to carry out its activities, and any equivalent costs imposed on other parties, including the regulated firms or individuals. They *do not include* what are sometimes called the real resource costs of the controls – that is, the costs of pollution control equipment, higher fuel bills for cleaner fuel, more expensive exhaust systems and so on. They also *do not include* any induced indirect costs that might occur such as loss of national competitiveness or increased unemployment. Summing up all those costs gives the total compliance costs of environmental regulation. Clearly, transactions costs are just one part – albeit a not insignificant part – of that overall total.

To clarify these ideas, it is helpful to look at Figure 7.11. We assume that the marginal gross benefits of pollution abatement (the damages avoided) are correctly represented by the curve labelled as D in the picture. Curve A represents the marginal real resource costs of pollution abatement. If there were no other costs, an efficient outcome would require Z_A units of abatement. There may also be induced, indirect costs, including impacts on unemployment and trade competitiveness. Adding these to the resource abatement costs, the composite cost curve B is obtained, with a correspondingly lower efficient abatement level, Z_B. Note that Figure 7.11 assumes that these induced indirect impacts are adverse to the economy in question. It is possible, though, that they may be beneficial. Double dividend effects could

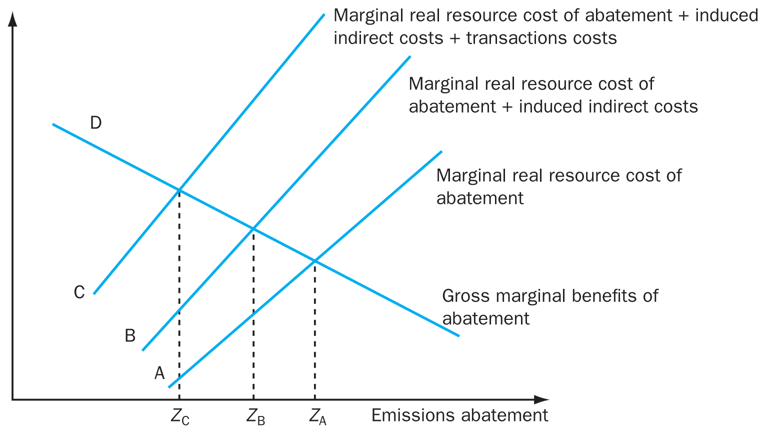


Figure 7.11 The net benefits of regulation

be interpreted as beneficial induced effects; moreover, there are reasons for believing that tighter environmental standards – particularly minimum technological requirements – might increase the dynamic growth potential of the economy. If the induced effects were beneficial rather than adverse overall, curve B would lie to the right (rather than to the left) of curve A.

Finally, curve C adds in transactions costs to the previous two categories of costs. The efficient abatement level, taking all relevant items of information into consideration, is Z_C. Figure 7.11 is intended only to illustrate and organise our thinking. Nevertheless, it does provide a useful way of thinking about instrument selection. Suppose that a choice of abatement target had already been made. That would then fix a particular point along the abatement axis that we are committed to reach. If we were comparing the relative merits of two instruments, we might construct two versions of Figure 7.11, one for each instrument. The preferred instrument would be the one that has the lower total cost of achieving that particular target. Even if one instrument is superior in terms of real resource cost of abatement, it need not be superior when induced effects and transactions costs are also considered.

Deviating for a moment from the main thrust of the argument, note that there is another interesting inference to be drawn from Figure 7.11. The notion that an EPA could devise an efficient target first, without consideration of instrument to be used, and then choose an instrument to best attain that target, may not be sensible. To see why, return to the idea that we might choose between instruments by constructing alternative versions of Figure 7.11, one for each instrument. It is evident that if the cost functions differ, so might the efficient abatement level. The independence of targets from instruments does not seem to go through. This should be intuitively clear after a little reflection, or by a slight change in our argument. Suppose that under existing technology instrument I is least-cost for achieving a target Z_I. Now suppose that an innovation creates a new instrument II, that is lower cost than I. It should be selected. But if it were selected, efficient abatement would now be higher, as some units of emission that could only be abated at a marginal cost exceeding the marginal benefit would now yield positive net benefit.

Let us now return to the issue of transactions costs. These arise, principally, because of uncertainty. The greater is that uncertainty, the larger will be these costs. In practice, these costs sometimes constitute a substantial proportion of the total costs of pollution control. One estimate of the size of transaction costs, for water quality control programmes in the USA, can be deduced from the information given in Box 7.1. It is evident that transaction costs are very large in that case. Moreover, it matters greatly how well the instrument referred to there compares with other control techniques that are available. Unfortunately, the source of information for Box 7.1 did not supply such information.

Box 7.1 US EPA estimates of the costs of the Clean Water TMDL programme

A recent draft report released by the US Environmental Protection Agency, in conjunction with a National Academy of Sciences study, estimated costs of the total maximum daily load (TMDL) programme, one tool used under the Clean Water Act for cleaning up United States rivers, lakes and streams. TMDLs are pollution limits set for a waterway, depending on its use. The limits are used to allocate any needed controls among all the pollutant sources, both point sources (industrial and municipal dischargers) and non-point sources (agriculture and urban runoff).

The programme could cost between $900 million and $4.3 billion dollars annually. These figures relate to programme abatement costs (that is, the costs of installing measures to reduce pollution) which would be borne primarily by dischargers, but do not include monitoring and compliance costs. These cost estimates include about 90 per cent of the waters currently on state lists for which the EPA currently has sufficient data to estimate clean-up costs. The EPA notes that the worst-case (but highly unlikely) scenario estimate of more than $4 billion to fully implement the clean-up is a fraction of current US expenditures for clean water.

In addition to these abatement costs, the EPA estimates the costs to states of additional data gathering to support the TMDL programme at $17 million per year. Once good data have been collected, states will need to spend up to $69 million annually over the next 15 years to develop plans to clean up some 20 000 impaired waters currently on state lists. State costs to develop a clean-up plan for each of these 20 000 waters are projected to average about $52 000 per plan. The costs quoted in this paragraph comprise some (but not all) of the programme's monitoring, compliance enforcement, and other transactions costs. Clearly, such costs are not negligible relative to the abatement costs.

Source: US EPA Press Release 3 August 2001. A copy of the report and additional information available online at: www.epa.gov/owow/tmdl

At this point there is little more that can be said *a priori* about optimal instrument choice. We have seen that total control costs might differ according to which kinds of instruments are used and also what kind of pollution problem is being considered. However, the magnitudes of transactions costs depend on prevailing circumstances, and probably have to be examined on a case-by-case basis. We have seen that economists sometimes have a presumption in favour of the use of incentive-based systems, as these can, under special circumstances, generate least-cost attainment of targets without knowledge of the cost structures of individual firms. However, arguments in this and the previous chapter have shown circumstances in which this line of reasoning may not be valid. For example, while the ambient permit system described in Chapter 6 is capable of being cost-efficient, it is likely to be very difficult or costly to design and implement where information is imperfect. Moreover, once established, it would not easily admit piecemeal changes that could be done using command and control. It is perhaps for these reasons that we rarely, if ever, find pure cases of ambient tradable permit schemes in practice.

When transactions costs are added in to relative cost comparisons, the cost-efficiency rankings of instruments may change, and it is far more difficult to reach general conclusions about the relative merits of different instruments. For example, required minimum technology standards are quick and simple to introduce, are relatively easy to monitor, and can be implemented in flexible ways. In other words, they generate low transactions costs. Standard arguments point to the likely cost-inefficiency of technology controls, which force firms to adopt a particular method of emissions reduction irrespective of whether it is the cheapest way in which that reduction could be achieved. The cost-inefficiency referred to here concerns the real resource costs of abating pollution. However, transaction cost advantages may outweigh that real resource cost disadvantage, and make technology controls a superior option to market-based instruments.

7.4.1 Regulatory failure

In this chapter and in Chapter 6, we have discussed the properties and relative advantages of various instruments that may be used to attain environmental policy targets. Much of our discussion has been premised on the assumption that efficiency losses can be minimised, or sometimes avoided entirely, by appropriate regulatory intervention.

Yet we have also noted that regulatory intervention is not costless. Regulation incurs a variety of transactions costs. These costs have to be set against the net benefits of regulation (the gross benefits of regulation minus the real resource costs of pollution abatement and minus any induced, indirect costs). In general, the presence of transactions costs will reduce the amount of pollution abatement that is warranted (see again Figure 7.11). In extreme cases, transactions costs may be sufficiently large to completely negate the expected net gains from regulation. Put another way, regulation costs exceed the attained benefits, and so regulation would lower – rather than increase – social net benefits. This situation could be described as one of regulatory failure to signify the fact that, as with market failure, social net benefit-maximising outcomes are not delivered.

But there are other causes of regulatory failure, two of which we finish our discussions with. First, regulatory action may also fail because of inadequate foresight or because of unintended consequences. Some interesting examples in the context of the United States Clean Air Act Amendments are outlined in Box 7.2. Second, regulatory failure may occur because of regulatory capture. This idea is outlined in Box 7.3. Finally, the reader may have noticed that our discussion has taken place under the implicit assumption that some authority has the ability to implement and administer a coordinated control programme. But many pollution problems spill over national boundaries, and cannot be properly dealt with by any single EPA. We discuss international coordination of environmental policy in Chapter 9.

Box 7.2 Regulatory failure in the case of the United States Clean Air Act Amendments of 1970 and 1977

Some interesting examples of regulatory failure arising from inadequate foresight and unintended consequences are provided by Joskow and Schmalensee (2000) in their discussion of the control of SO_2 emissions from large electricity utility sources. The 1970 Clean Air Act Amendments (CAAA) established a system of national maximum ambient air quality standards, and gave states the principal responsibility to ensure compliance with those standards. To meet local standards, some states imposed minimum stack height requirements. While contributing to local clean-up, this did not reduce total emissions. Indeed, it almost certainly led to increased acid rain deposition (over a much wider area), as SO_2 at higher levels of the atmosphere persists longer and is a more effective precursor of acid rain formation. A second feature of the 1970 CAAA was the 'new source performance standards' (NSPS) for newly commissioned generating sources. NSPS set maximum emissions *rates*, in terms of mass of SO_2 per mass of fuel burned. These regulations created a substantial gap in allowable emissions rates between old and new plant, and so provided a powerful incentive to prolong the life of old (and dirtier) plant.

The 1977 CAAA extended the scope of emissions control and required that coal-fired plant built after 1978 meet both the 1970 emission rate requirement and either

(a) to remove 90% or more of potential SO_2 emissions, or
(b) to remove 70% or more of potential SO_2 emissions and to operate with an emissions rate no more than half of the 1970 requirement.

This was widely interpreted as a major tightening of emissions control, and was welcomed by environmentalists. However, the 'percentage reduction' component of the legislation required all generators to use flue gas desulphurisation equipment (scrubbers) *even if* they already used low-sulphur coal. This had the consequence of largely removing the cost advantage of low-sulphur coal as means of compliance. However, drawing on some observations made by Ackerman and Hassler (1981), the authors note that in spite of there now being stricter emissions limits on new sources, the 1977 CAAA probably led to greater overall air pollution by encouraging utilities to burn high-sulphur coal and by prolonging the life of old generating plant.

Box 7.3 Regulatory capture

The notion of regulatory capture is one aspect of principal–agent theory. It refers to the idea that a regulator, entrusted with the task of shaping the behaviour of economic agents so as to achieve various stated public-policy objectives, may find itself 'captured' by those whom it is supposed to regulate, and effectively becomes a means by which the latter regulate themselves in a self-serving manner.

At one level, regulatory capture is an assertion about what actually exists, or has existed at various times and places. It can also be thought of as a hypothesis about what is possible under various structures of incentives and relationships, and as a guide to how a regulatory regime might be designed to minimise the chances of those outcomes occurring.

The theoretical foundation for regulatory capture can be found in the so-called 'new political economy'. Here the policy maker is seen as making rational (optimising) choices subject to patterns of incentives and networks of pressures (Arrow, 1951; Olson, 1965). Several schools of thought have emerged within this framework, including public choice theory, in which political behaviour reflects maximisation of the probability of electoral success (Buchanan and Tullock, 1962) and the political economy of regulation (Stigler, 1971; Peltzman, 1976).

A key component of the latter is the capture theory of regulation. The major principles underlying regulatory capture can be summarised as follows. Actual or potential regulated parties (firms, let's say) may be adversely affected by regulation. Where they do stand to lose from regulatory action, firms have incentives to avert those adverse consequences. To do so, firms may be able to bring to bear on policy makers or policy administrators a variety of pressures and influences (to be outlined below). Of course, regulation in principle offers the prospect of benefiting the general public. Therefore one might expect countervailing pressure to be brought to bear by the public through electoral and other political processes.

However, the relatively small numbers of firms means that their cost of organising to bring about pressure are small relative to those of the general public. Moreover, each individual regulated party will usually have much more to lose than an individual citizen, and so has a stronger incentive to incur costs to head off that regulation.

By what means can potentially regulated firms exert pressure to manipulate policy or its implementation in their favour? There are several, including:

- Lobbying pressure in the policy-making and legislative processes.
- Funds for supporting candidates in elections or in their search for administrative office: financial contributions may buy support for future influence.
- Long-term relationships between regulators and regulated. In many command and control regimes, regulation is highly decentralised, and is reliant on flows of information that derive from continuous working relationships. In these circumstances, regulators will tend to become increasingly imbued with the corporate ethos and culture of the regulated parties.
- Revolving-door career profiles. In some cases, those in mid-career as agents of regulatory agencies secure senior positions in firms within the regulated sector, acting as directors, advisers or lobbyists. Where these career patterns become common and expected by both sides, the independence of regulators may be compromised. Sanjour (1992), safeguarded in employment in the US EPA by whistle-blower protection, provides an insider account of revolving-door career profiles and other reciprocal influence relationship positions between the EPA and the hazardous waste industry.

In the final analysis, the extent to which regulatory capture actually takes place is an empirical question. However, it is clear from these brief comments that, to the extent it does take place, regulatory capture partly involves influence at the policy- or law-making stage, and partly in the processes by which by which laws are implemented and administered. Downing (1981) provides an illuminating model of the implementation of pollution control legislation. Three groups – polluters, victims of pollution, and the regulator – participate in this regulatory game. The environmental protection agency seeks to maximise some function of three underlying objectives: environmental improvement, increased agency budget, and its discretionary power. Each of these objectives gives opportunities for networks of influence to be established.

Firms will not necessarily act in a unified way, of course. In general, each sector, region or firm is out to get as much as it can from the implementation process, and so to some degree

Box 7.3 *continued*

a zero-sum game is being played out. Nor does regulatory capture necessarily reduce the extent of regulation. Milliman and Prince (1989) discuss the incentives of firms to innovate, and show that those firms that succeed in developing low-cost control innovations may lobby for greater regulation in order to gain competitive advantage. Maxwell *et al*. (1996) demonstrate that polluters may sometimes have incentive to undertake limited voluntary clean-up to head off the likelihood of pollution victims organising to press for stricter regulatory control.

Joskow and Schmalensee (2000) provide a detailed case study of the political economy of US SO_2 control and acid rain legislation. They argue that the 1970 and 1977 Clean Air Act legislation was

> an excellent example of interest group politics mediated through legislative and regulatory processes. . . . Concentrated and well-organized interests in a few states that produced and burned high-sulfur coal were able to shape the Clean Air Act Amendments (*CAAA*) of 1970 and, particularly, 1977 to protect high-sulfur coal and impose unnecessary costs on large portions of the rest of the country.

Even with the 1990 CAAA, those states with powerfully organised Congressional presence were able to secure substantial benefits in their allocations of emissions allowances for Phase I units (the class of old and dirty electricity generators). However, the authors note that allocations for Phase II units were not consistent with simple interest-group models:

> If anything, the resulting allocation of Phase II allowances appears more to be a majoritarian equilibrium than one heavily weighted towards a narrowly defined set of economic or geographical interests.
>
> *pp. 642–643*

Summary

- Whatever criterion is used, or objective sought, setting pollution targets and choosing among pollution control instruments is made difficult by uncertainty.
- In many circumstances, the EPA will not only be unaware of the abatement costs of individual firms but it will not even know the aggregate emissions abatement cost function. It will be difficult or impossible to determine efficient targets in those circumstances.
- In a world of certainty, where the EPA knows all relevant information, no instrument has an advantage over any other in cost-efficiency terms, provided that transactions costs do not differ among instruments.
- If transactions costs do exist, and they differ from one type of instrument to another, this can be important in selection of an appropriate pollution control instrument.
- Under uncertainty, instruments differ in cost-efficiency properties.
- For uniformly mixing pollutants, economic incentive instruments have the important advantage over command and control that the EPA needs to know less in order to attain targets cost-effectively. Specifically, it does not require knowledge of individual firms' abatement costs. In contrast, an emissions tax, abatement subsidy or marketable permit scheme can be used to achieve a predetermined emissions target at least total cost with the EPA knowing only the aggregate abatement cost schedule.
- Even if the EPA did not know the aggregate abatement cost schedule, use of any of these economic-incentive instruments would achieve some target at least cost.
- Where the pollutant is not uniformly mixing, two of the three incentive-based instruments – emissions taxes and abatement subsidies – lose their relative advantage over command and control licences. Only marketable permits (in the form of an ambient permit system described in

Section 7.5) allow the EPA to reach pollution targets at least cost in the absence of perfect knowledge about firms' costs.

- For linear functions, when the EPA has uncertainty about the marginal abatement cost function, it should prefer a quantity policy (licences) to an emissions tax if MC is flatter than MD, and an emissions tax to a licence system if the reverse is true, if it wishes to minimise the efficiency losses arising from incorrect information.
- While economists attach great importance to efficiency and optimality as criteria in setting policy targets, operationalising these criteria is often very difficult because of limited information and uncertainty.
- In the presence of uncertainty, it may be appropriate for pollution policy to take some account of the precautionary principle. This is likely to be particularly appropriate where uncertainty is acute, damage effects are thought to be catastrophic, and non-linearities may be present in pollution–damage relationships.

Further reading

Baumol and Oates (1988) is a classic source in the area of environmental regulation. The whole book is relevant but it is quite difficult and formal. Hartwick and Olewiler (1998) is less difficult, and contains some good expositions of policy making with imperfect information.

Two seminal works on the new political economy are Arrow (1951) and Olson (1965). The classic work in public choice theory (in which political behaviour reflects maximisation of probability of electoral success) is Buchanan and Tullock (1962). Important original works in the political economy of regulation are Stigler (1971) and Peltzman (1976). Sanjour (1992) gives an insider's account of EPA regulatory failure in relation to the US EPA and the hazardous waste industry. Downing (1981) develops a political economy model of the implementation of pollution control legislation. Milliman and Prince (1989), Salop *et al*. (1984) and Hackett (1995) analyse the incentives to innovate in abatement technology. Maxwell *et al*. (1996) argue that polluters have incentive to undertake limited voluntary clean-up to head off victims organising to press for stricter regulatory control. Joskow and Schmalensee (2000) provide a detailed analysis of the political economy of regulation for the case of the US acid rain programme. The EPA website (at May 2002) contains an interesting set of documents relating to whether or not environmental regulation is effective and beneficial. Goodstein (1999) provides some interesting evidence regarding the 'revolving-door' effect. Government failure is the central theme of Weimer and Vining (1992).

Discussion of the idea of a safe minimum standard of conservation can be found in Bishop (1978) and Randall and Farmer (1995). Stebbing (1992) discusses the notion of a precautionary principle applied to pollution policy. A number of texts provide collections of papers, several of which are relevant to pollution control policy under imperfect information; these include Bromley (1995) and, at a more rigorous level, the three 'Handbooks' edited by Kneese and Sweeney (1985a, 1985b, 1993).

Discussion question

1. Asymmetric information typically involves the regulator having less relevant information than the regulated parties. Find out what is meant by 'adverse selection' and show why it can lead to asymmetric information. Why does adverse selection make it difficult to regulate pollution efficiently?

Problems

1. If the control authority does not know the marginal damage function, it will not be able to identify the economically efficient level of pollution abatement, nor the efficient tax or subsidy level. Demonstrate that
   (a) knowledge of the pollution abatement cost schedule alone means that it can calculate the required rate of tax to achieve any target level it wishes;
   (b) if the regulator knew neither the marginal damage nor the marginal abatement cost schedules, then it could arbitrarily set a tax rate, confident in the knowledge that whatever level of abatement this would generate would be attained at minimum feasible cost.

2. We examined in this chapter a particular form of asymmetric information where a firm knows its emissions abatement costs but the regulator does not. The regulator must ask firms to reveal abatement costs in order to select an efficient emission reduction target. We demonstrated that where a firm expects the EPA to use a marketable permit system, it may be in the interest of firms to exaggerate their marginal abatement costs (MC). In contrast, where a firm expects the EPA to use an emissions tax, firms have an incentive to understate their abatement costs.

   Suppose that the regulator committed itself to randomly selecting either a fee or a marketable permit system (each with a probability of one-half), but only after it received cost information from firms. Would this system generate truthful reporting by firms?

3. A regulator requires a company to reduce its emissions to a level below its (pre-regulation) profit-maximising level of emissions. The requirement will be implemented by the issue of a non-transferable licence. However, imperfect monitoring means that if the firm does not adhere to the regulation, the probability of this being discovered is p, which will in general be significantly less than one. If the company is discovered to have not adhered to the licence, however, it will face a financial penalty of £x per unit emitted in excess of its allowed (licensed) amount.

   Show how the amount which it is optimal for your company to emit (i.e. the amount which will maximise your expected profits) depends on the values of p and x. Would the company's decision be different if the penalty were a fixed fine, irrespective of the magnitude of its transgression?

4. You are given the following information:
   (a) A programme of air pollution control would reduce deaths from cancer from 1 in 8000 to 1 in 10 000 of the population.
   (b) The cost of the programme is expected to lie in the interval £2 billion (£2000 million) to £3 billion annually.
   (c) The size of the relevant population is 50 million persons.
   (d) The 'statistical value' of a human life is agreed to lie in the interval £300 000 to £5 million.

   If the only benefit from the programme is the reduced risk of death from cancer, can the adoption of the programme be justified using an economic efficiency criterion?

CHAPTER 8 Economy-wide modelling

A model is simply an ordered set of assumptions about a complex system. . . . The model we have constructed is, like every other model imperfect, oversimplified, and unfinished.

Meadows *et al.* (1972), p. 20

Learning objectives

In this chapter you will

- learn about the basic input–output model of an economy and its solution
- find out how the basic input–output model can be extended to incorporate economy–environment interactions
- encounter some examples of environmental input–output models and their application
- learn how the input–output models, specified in terms of physical or constant-value flows, can be reformulated to analyse the cost and price implications of environmental policies, such as pollution taxes, and how these results can be used to investigate the distributional implications of such policies
- study the nature of computable general equilibrium (CGE) models and their application to environmental problems

Introduction

Appropriate environmental policy measures require a detailed understanding of the environmental impact of particular economic activities, and hence there is the need to model the relationships between the economy and the environment. For instance, which economic activities result in the emission of carbon dioxide, and by how much would particular economic activity levels have to be reduced to bring about a reduction of, say, 20% in CO_2 emissions? What level of 'carbon tax' would be necessary to bring about such a reduction? What would be the effects of such a tax on different types of household? For many policy purposes it is not enough to know simply the nature and direction of the changes that would be brought about by a particular measure (or by the failure to implement a measure) – a quantitative estimate of the effects of the policy (or of its absence) is needed. It is for this purpose that empirical models of interaction between the economy and the environment are constructed.

By using models to assess and compare the simulated quantitative effects of a range of feasible policy options, governments can hope to identify the 'best' (or least bad) policy or policy mix, avoid policy combinations that are inconsistent or that work in opposite directions, and achieve some kind of optimal trade-off between different, and potentially conflicting, economic and environmental objectives.[1] Moreover, such simulation exercises underpin the

[1] Of course, what is optimal to a government may not seem optimal to some interest groups, such as environmentalists, or the unemployed, or the political opposition, each of whom may, and most probably will, have different social welfare functions (or different perceptions of the social welfare function), and will attach different weights to particular economic and environmental outcomes.

formulation and implementation of proactive environmental policies, which attempt to anticipate or avoid undesirable outcomes by appropriate preventive measures. Although formal simulation modelling is not a precondition for proactive environmental policies, it can powerfully influence public attitudes and policy making, a recent example being predictions and simulations of the effects of greenhouse gases on global warming. In the absence of quantitative models of economy–environment interaction, policy is more likely to be reactive than proactive, and may be too late if environmental damage is irreversible (for example, species extinction).

The usefulness of such model simulations is greatly enhanced where the models are multi-sectoral, distinguishing different commodities in consumption and production. The production and use of some commodities has very little environmental impact, whereas with other commodities the impact is large. Fossil fuels are examples of the latter kind, and models which enable the analyst to consider, for example, the effects of technological change in the energy sector are very useful. Box 8.1 reports on the use of a multi-sectoral model of the world economy to examine the feasibility of sustainable development as envisaged in the Brundtland Report.

Several types of multi-sectoral model have been used to examine economy–environment interactions: input–output models, computable general equilibrium models, and linear and non-linear programming (optimisation) models. This chapter is largely devoted to a discussion of environmental input–output (I/O) models and their application; these have been used quite extensively in environmental

Box 8.1 Using input–output analysis to consider the feasibility of sustainable development

The Brundtland Report (see Chapter 2 here) claimed that sustainable development was feasible, asserting that given the political will and the necessary institutional changes the world economy could grow fast enough to significantly reduce poverty without increasing environmental damage. This was an assertion rather than a demonstration, in as much as the report did not put together the technological and economic possibilities looked at in various parts of the report and examine them for consistency. Duchin and Lange (1994), hereafter DL, is a report on a multi-sector economic modelling exercise, using input–output analysis to look at the feasibility of sustainable development as envisaged in the Brundtland Report.

DL used an input–output model of the world economy which distinguished 16 regional economies, in each of which was represented the technology of 50 industrial sectors. This model was used to generate two scenarios for the world economy for the period 1990 to 2020. The reference scenario assumes that over this period world GDP grows at 2.8% per year, while the global population increases by 53%. DL take 2.8% per year to be what is implied by the Brundtland Report's account of what is necessary for sustainable development. In this reference scenario production technologies are unchanging over the period 1990–2020.

The second scenario is the OCF scenario, where OCF stands for Our Common Future, the title of the Brundtland Report. This uses the same global economic and demographic assumptions as the reference scenario, but also has technologies changing over 1990 to 2020. In the OCF scenario DL incorporate into the input–output model's coefficients technological improvements as envisaged in the Brundtland Report in energy and materials conservation, changes in the fuel mix for electricity generation, and measures to reduce SO_2 and NO_2 emissions per unit energy use.

As indicators of environmental impact, the analysis uses the input–output model to track fossil fuel use and emissions of CO_2, SO_2 and NO_2. In the reference scenario, all of these indicators increase by about 150% over 1990 to 2020. With the technological changes, there are big environmental improvements in the OCF scenario. But the indicators still go up – by 61% for fossil fuel use, by 60% for CO_2, by 16% for SO_2, and by 63% for NO_2. Given the assumed economic and population growth, the technological improvements are not enough to keep these environmental damage indicators constant. DL conclude that sustainable development as envisaged in the Brundtland Report is not feasible. As DL put it, 'the position taken in the Brundtland Report is not realistic' (p. 5) in as much as 'the economic and environmental objectives of the Brundtland Report cannot be achieved simultaneously' (p. 8).

economics, particularly in studies related to energy and pollution. They are the basis for computable general equilibrium (CGE) models, which will also be discussed. It must be stressed that the availability and quality of data, both economic and environmental, are serious impediments to the development of all kinds of models of links between the economy and the environment.

The following section presents and explains the basic input–output model and its solution, while the next section shows how the basic model can be extended to incorporate economy–environment interactions, and includes examples of environmental input–output models and their application. These applications are concerned with the 'real' side of the economy, that is, with physical or constant-value flows. We then show in Section 8.3 how the equations of the model can be reformulated to analyse the cost and price implications of environmental policies, such as pollution taxes, and how these results can be used to investigate the distributional implications of such policies. The last section of the chapter reviews the nature of CGE models and their application to environmental problems. The first appendix makes extensive use of matrix algebra to present a very general framework for environmental input–output analysis, while the second works through the algebra of the simple two-sector CGE model used in the final section of the chapter.

8.1 Input–output analysis

I/O models incorporate a number of simplifying assumptions which require a degree of caution in interpreting their results, but they are mathematically tractable and less demanding of data than many other multisectoral models. The basis for input–output modelling is the availability of suitable economic data, and we begin with a discussion of the accounting conventions according to which such data are made available.

8.1.1 Input–output accounting

The basis of the input–output system is the transactions table, which is essentially an extended version of the national accounts in which inter-industry transactions – that is, flows of goods and services between industries – are explicitly included and indeed form the centrepiece of the system of accounts. This contrasts with the conventional national accounts in which inter-industry transactions are 'netted out', and the accounts record only the value added by each industry, and the value of sales to final buyers.

Table 8.1 is a hypothetical example of a transactions table, in which all production activities in the economy have been allocated to one of three sectors. Looking across any row of the table shows what the sector on the left sold to each sector at the top, for example:

$$\underset{\text{(to)}}{\text{Agriculture sales}} = \underset{\text{(Agriculture)}}{0} + \underset{\text{(Manufacturing)}}{400} + \underset{\text{(Services)}}{0} + \underset{\text{(Households)}}{500} + \underset{\text{(Exports)}}{100} = \underset{\text{(Total output)}}{1000}$$

Table 8.1 Input–output transactions table, $ million

	Sales to:	Intermediate sectors			Final demand		
	Purchases from	Agriculture	Manufacturing	Services	Households	Exports	Total output
Intermediate sectors	Agriculture	0	400	0	500	100	1000
	Manufacturing	350	0	150	800	700	2000
	Services	100	200	0	300	0	600
Primary inputs	Imports	250	600	50			
	Wages	200	500	300			
	Other value added	100	300	100			
	Total input	1000	2000	600			

Notice that sales are divided between those to intermediate sectors (agriculture, manufacturing and services) and to final demand (households and exports).[2]

The sum of intermediate and final sales for each sector is gross, or total, output. Again, for simplicity, we assume no government or investment expenditure, which normally would be included as additional components of final demand. Looking down any column of the table shows what the sector listed at the top purchased from each sector on the left, for example:

$$\text{Manufacturing purchases (from)} = \underset{\text{(Agriculture)}}{400} + \underset{\text{(Manufacturing)}}{0} + \underset{\text{(Services)}}{200} + \underset{\text{(Imports)}}{600} + \underset{\text{(OVA)}}{300} = \underset{\text{(Total input)}}{2000}$$

Notice that purchases are divided between those from intermediate sectors (agriculture, manufacturing and services), and so-called 'primary input' purchases (imports, wages and other value added).

Transactions tables normally account for flows over a period of one year. They are also typically expressed in value terms, in order to provide a standard unit of account across sectors, though in principle it would be possible to use sector-specific units of account (tonnes, metres, numbers, therms), or a combination of physical and monetary units. A real transactions table will normally be larger than Table 8.1 because more sectors will be separately identified but the interpretation of it will be the same. A recently compiled input–output table for the UK, for example, contains 123 intermediate sectors, and the most recent table for the United States has 480 intermediate sectors. Tables of this size provide a highly detailed snapshot of the structure of an economy in a particular year, and show the pervasive interdependence of sectors and agents. Because their compilation requires the gathering and processing of a very large amount of data, there is often a long period between the year to which an actual input–output table relates and the year of its first publication.[3]

Because of the accounting conventions adopted in the construction of an I/O transactions table, the following will always be true:

1. For each industry: Total output ≡ Total input, that is, the sum of the elements in any row is equal to the sum of the elements in the corresponding column.
2. For the table as a whole: Total intermediate sales ≡ Total intermediate purchases, and Total final demand ≡ Total primary input

Note the use here of the identity sign, ≡, reflecting the fact that these are accounting identities, which always hold in an I/O transactions table.

The standard national income accounts can be readily derived from the input–output accounts. For example, GDP can be derived from Table 8.1.

1. On the Income side as:

Wages	\$1000m
+ OVA	\$500m
= GDP	\$1500m

or

2. On the Expenditure side as:

Household expenditure	\$1600m
+ Exports	\$800m
– Imports	\$900m
= GDP	\$1500m

Reading across rows the necessary equality of total output with the sum of its uses for each industry or sector can be written as a set of 'balance equations':

$$X_i \equiv \Sigma_j X_{ij} + Y_i,\ i = 1, \ldots, n \qquad (8.1)$$

where X_i = total output of industry i
X_{ij} = sales of commodity i to industry j
Y_i = sales of commodity i to final demand
n = the number of industries (3 in Table 8.1)

[2] As a further simplification, transactions between undertakings within the same sector (intra-industry transactions) have been netted out, so that the main diagonal of Table 8.1 has zeros everywhere.

[3] The details of the methods for the compilation and publication of input–output data vary across countries, and are described in publications by national statistical agencies. For a short recent account of UK practice see Mahajan (2006).

8.1.2 Input–output modelling

To go from accounting to analysis, the basic input–output modelling assumption is that

$$X_{ij} = a_{ij}X_j \tag{8.2}$$

where a_{ij} is a constant. That is, it is assumed that intermediate inputs are constant proportions of the output of the purchasing industry. So for example if X_j represents the output of the steel industry (tonnes valued at constant prices) and X_{ij} records purchases of iron ore (tonnes valued at constant prices) by the steel industry, we are assuming that iron ore purchases are a constant fraction of the value of steel output (expressed in constant prices); if the output of steel doubles, inputs (purchases) of iron ore will double.

Substituting equation 8.2 into 8.1 gives

$$X_i = \Sigma_j a_{ij}X_j + Y_i,\ i = 1, \ldots, n \tag{8.3}$$

as a system of n linear equations in $2n$ variables, the X_i and Y_i, and n^2 coefficients, the a_{ij}. If the Y_i – the final demand levels – are specified, there are n unknown X_i – the gross output levels – which can be solved for using the n equations. Given that the equations are linear, the solution can readily be accomplished using matrix algebra.[4] In matrix notation, equations 8.3 become

$$\mathbf{X} = \mathbf{AX} + \mathbf{Y}$$

which on rearrangement is

$$\mathbf{X} - \mathbf{AX} = \mathbf{Y} \tag{8.4}$$

where $\mathbf{X}$ is an $n \times 1$ vector of gross outputs, X_i, $\mathbf{A}$ is an $n \times n$ matrix of intermediate input coefficients, a_{ij}, and $\mathbf{Y}$ is an $n \times 1$ vector of final demands, Y_i.

With $\mathbf{I}$ as an $n \times n$ identity matrix, equation 8.4 can be written as

$$(\mathbf{I} - \mathbf{A})\mathbf{X} = \mathbf{Y}$$

which has the solution

$$\mathbf{X} = (\mathbf{I} - \mathbf{A})^{-1}\mathbf{Y} \tag{8.5}$$

where $(\mathbf{I} - \mathbf{A})^{-1}$ is the 'inverse' of $(\mathbf{I} - \mathbf{A})$. This solution can be written as

$$\mathbf{X} = \mathbf{LY} \tag{8.6}$$

where $\mathbf{L} = (\mathbf{I} - \mathbf{A})^{-1}$, and the notation $\mathbf{L}$ used as this inverse is often referred to as the 'Leontief inverse' in recognition of the progenitor of input–output analysis, Wassily Leontief.

This is the basic input–output model. Its use involves a number of assumptions – notably equation 8.2 and the constancy of the a_{ij} – which are clearly approximations to reality, but used judiciously input–output modelling can be a cost-effective and powerful tool in a number of applications. In the next section it is shown how the basic model can be extended to incorporate economy–environment interactions. Before doing that, it will be useful to consider further what it is that the basic model does, and to work through a numerical illustration of its application based on Table 8.1. For the reader who wishes to verify the calculations, or who wishes to see how Excel can be used to do the kinds of matrix algebra calculations used in input–output analysis, the numerical example is reproduced in its entirety in an Excel file on the companion website in the *Additional Materials* for Chapter 8, *IO.xls*.

For a three-sector economy, the matrix equation 8.6 can be written in ordinary algebra as the three equations

$$\begin{aligned} X_1 &= l_{11}Y_1 + l_{12}Y_2 + l_{13}Y_3 \\ X_2 &= l_{21}Y_1 + l_{22}Y_2 + l_{23}Y_3 \\ X_3 &= l_{31}Y_1 + l_{32}Y_2 + l_{33}Y_3 \end{aligned} \tag{8.7}$$

where the l_{ij} are the elements of the Leontief inverse, $\mathbf{L}$. Each equation here gives the gross output of an industry as depending on the levels of final demand for each of the three commodities. The l_{ij} give the level of output in the ith industry to meet the direct and indirect requirements for a unit of final demand for commodity j. Thus, for example, the delivery of one unit of commodity 2 to final demand requires an output level l_{22} in industry 2. This level will meet both the direct requirement and the indirect requirement arising from the fact that commodity 2 is used in the production of commodities 1 and 3, which are used in the production of commodity 2. The actual

4 Readers who are unfamiliar with matrix algebra will find the essentials in Appendix 5.1 to Chapter 5.

gross output requirement for commodity 2, X_2, depends, in the same way, on the levels of delivery of all three commodities to final demand, as given by l_{21}, l_{22} and l_{23}, and similarly for X_1 and X_3.

From the data in Table 8.1, and using equation 8.2, the elements of the matrix **A** are calculated as follows:

$$a_{11} = 0,\ a_{12} = 400/2000 = 0.2000,\ a_{13} = 0$$

$$a_{21} = 350/1000 = 0.3500,\ a_{22} = 0,\ a_{23} = 150/600 = 0.2500$$

$$a_{31} = 100/1000 = 0.1000,\ a_{32} = 200/2000 = 0.1000,\ a_{33} = 0$$

Hence,

$$\mathbf{A} = \begin{bmatrix} 0.0000 & 0.2000 & 0.0000 \\ 0.3500 & 0.0000 & 0.2500 \\ 0.1000 & 0.1000 & 0.0000 \end{bmatrix}$$

and

$$\mathbf{I} - \mathbf{A} = \begin{bmatrix} 1.0000 & -0.2000 & 0.0000 \\ -0.3500 & 1.0000 & -0.2500 \\ -0.1000 & -0.1000 & 1.0000 \end{bmatrix}$$

so that[5]

$$\mathbf{L} = (\mathbf{I} - \mathbf{A})^{-1} = \begin{bmatrix} 1.0833 & 0.2222 & 0.0556 \\ 0.4167 & 1.1111 & 0.2778 \\ 0.1500 & 0.1333 & 1.0333 \end{bmatrix}$$

Substituting into equation 8.6 gives

$$\mathbf{X} = \begin{bmatrix} 1.0833 & 0.2222 & 0.0556 \\ 0.4167 & 1.1111 & 0.2778 \\ 0.1500 & 0.1333 & 1.0333 \end{bmatrix} \begin{bmatrix} Y_1 \\ Y_2 \\ Y_3 \end{bmatrix}$$

which is the same as:

$$X_1 = 1.0833Y_1 + 0.2222Y_2 + 0.0556Y_3$$

$$X_2 = 0.4167Y_1 + 1.1111Y_2 + 0.2778Y_3$$

$$X_3 = 0.1500Y_1 + 0.1333Y_2 + 1.0333Y_3$$

If the Y_i here are replaced by the total final demand levels from Table 8.1 ($Y_1 = 600$ for agriculture, $Y_2 = 1500$ for manufacturing, $Y_3 = 300$ for services) this gives the gross output levels

Agriculture	$X_1 = 999.96$
Manufacturing	$X_2 = 2000.01$
Services	$X_3 = 599.94$

which are the same, allowing for inevitable small errors on account of rounding, as the total output levels shown in Table 8.1.[6] This must be the case, given that the final demand levels are the same.

Now suppose that it is known that there will be an increase in the export demand for all commodities. What are the implications for gross output levels? Suppose that the new export demand levels are

Agriculture	200
Manufacturing	1000
Services	100

so that the new final demand levels are:

$$Y_1 = 700$$
$$Y_2 = 1800$$
$$Y_3 = 400$$

Using these with the elements of the Leontief inverse, as above, gives as the new gross output levels:

$$X_1 = (1.0833 \times 700) + (0.2222 \times 1800) + (0.0556 \times 400) = 1180.51$$

$$X_2 = (0.4167 \times 700) + (1.1111 \times 1800) + (0.2778 \times 400) = 2402.79$$

$$X_3 = (0.1500 \times 700) + (0.1333 \times 1800) + (0.0333 \times 400) = 758.26$$

Note that for every industry the increase in gross output exceeds the increase in the final demand for the commodity that it produces. This is because commodities are used in the production of commodities. Input–output analysis is the investigation of the quantitative implications of such inter-industry relations.

[5] While the inverse for a small matrix can be found using a calculating machine, using a method described in, for example, Chiang (1984), the calculation is tedious and prone to error. Even for small matrices, it is better to use the routine included in most spreadsheet packages for PCs. One such routine is explained in the Word document *Matrix.doc* and illustrated in the Excel file *Matrix.xls*. These are both available from the *Additional Materials* for Chapter 5.

[6] This calculation, and others in this chapter, were done with a pocket calculator. If an Excel spreadsheet is used, as in the file *Matrix.xls* in the *Additional Materials* to Chapter 6, a higher degree of accuracy will be obtained.

8.2 Environmental input–output analysis

Proposals to extend input–output tables and models to include aspects of economy–environment links were first mooted in the late 1960s. The next 10–15 years saw a rapid development of environmental input–output models. Although there are some important differences between the models developed by different authors – so that for particular applications the choice of model is important – they all share a common basis of input–output methodology, including constant returns to scale production functions which permit no substitution between inputs (Leontief production functions), as described in the preceding section. A general input–output framework for economy–environment linkages is presented, using matrix algebra, in Appendix 8.1. Here we consider some particular, but useful and widely used, forms of economy–environment analysis using input–output accounting and modelling.

8.2.1 Analysing the effect of final demand changes

Suppose that in addition to the data of Table 8.1 we also know that the use of oil by the three industries was

Agriculture	Manufacturing	Services
50	400	60

where the units are petajoules, PJs. A joule is a unit of measurement used in recording the use of energy in the economy (one joule is the energy conveyed by one watt of power for one second) and the prefix 'peta' stands for 10^{15}. With O_i for oil use by the ith industry, paralleling equation 8.2, assume

$$O_i = r_i X_i \tag{8.8}$$

so that

$r_1 = 0.05$ for agriculture
$r_2 = 0.2$ for manufacturing
$r_3 = 0.1$ for services

These coefficients can be used to figure the implications for total oil use of changes in deliveries to final demand by applying them to the changes in the X_i associated with the change in final demand. Thus, for example, in the previous section the following changes to final demand deliveries

$$\Delta Y_1 = 100 \quad \Delta Y_2 = 300 \quad \Delta Y_3 = 100$$

were found to imply

$$\Delta X_1 = 180.51 \quad \Delta X_2 = 402.79 \quad \Delta X_3 = 158.26$$

so that the oil use changes are

$$\Delta O_1 = 9.03 \quad \Delta O_2 = 80.56 \quad \Delta O_3 = 15.83$$

with an increase in total oil use from 510 to 615.41 PJ.

A similar approach can be used in regard to waste emissions. If the emissions levels of a particular kind, E_i, are known, then, paralleling equation 8.8, assume

$$E_i = w_i X_i \tag{8.9}$$

and the implications of final demand changes for these emissions can be figured as just described for oil. Clearly, the same procedure can be followed for any number of particular kinds of emissions (or resource inputs), if the data are available.

An early example of the application of this kind of analysis is McNicoll and Blackmore (1993). This study calculated emissions coefficients for 12 pollutants for a (preliminary) 29-sector version of the 1989 input–output tables for Scotland. Coefficients were expressed in tonnes per £ million output, except radioactivity, which is measured in thousand becquerels. Applications of the model included a number of simulation studies, two of which involved assessing the impact on pollution emissions of (i) partial substitution by consumers of coal by gas, and (ii) partial substitution for road and air transport by rail transport. For SIM1 (coal by gas), final demand for coal was reduced by £30m, while that for gas was increased by £30m. For SIM2 (greater use of rail), final demand for road transport was reduced by £50m, and air transport by £20m, while final demand for rail transport was raised by £70m. In both cases aggregate final demand was kept unchanged in order to show the effects of different patterns of expenditure. Although the figures used are purely illustrative, the approach and discussion are suggestive of how environmental input–output models can be used to

Table 8.2 SIM1 and SIM2 impacts on outputs and emissions

	Δ Gross output[a]			Δ Emissions[b]	
Sector	SIM1	SIM2	Emission	SIM1	SIM2
1	−0.04	+0.64	CO_2	−404.7	−287.7
2	−0.001	−0.04	CO_2(weight)	−110.4	−78.5
3	−0.02	+0.22	SO_2	−3.0	+0.11
4	−30.21	+0.08	Black smoke	−0.35	−0.06
5	−0.02	−0.02	NO_x	−0.88	−3.84
6	−0.73	−1.06	VOC	−0.14	−3.25
7	−0.27	+2.25	CO	−0.35	−21.16
8	+30.98	+0.60	Methane	−11.46	+0.09
9	−0.01	+0.17	Waste	−655.2	+24.26
10	−0.11	+0.07	Lead	−0.000005	−0.01
11	+0.03	+1.02	RA air	−0.001	+0.009
12	+0.004	+0.27	RA water	−0.00006	+0.0005
13	−0.09	+0.21	RA solid	0.0143	+0.121
14	−0.15	+0.14			
15	−0.009	+1.20			
16	−0.19	+2.13			
17	+0.02	+0.83			
18	+0.05	+0.97			
19	−0.08	+0.48			
20	+0.01	+1.65			
21	−0.42	+15.25			
22	+0.02	+70.29			
23	−0.04	−49.2			
24	+0.01	+0.21			
25	+0.09	−20.49			
26	+0.61	+2.80			
27	+2.64	+5.47			
28	−0.24	+2.89			
House	−5.84	+71.47			

Notes: [a] Units are £ × 10^6
[b] Units are tonnes × 10^3, except for RA
Source: Adapted from McNicoll and Blackmore (1993)

quantify and evaluate the effects of policies which influence the pattern, as well as the level, of economic activity. Especially in SIM2, for example, it is interesting that although rail travel is usually considered more environmentally friendly than road or air transport, the substitution suggests an increase in the output of certain pollutants.

Results of the simulations are summarised in Table 8.2. The left-hand columns record the estimated effects of the substitutions on sector outputs, compared with actual 1989 outputs. The right-hand columns show the estimated changes in emissions which would result from the substitutions. For SIM1, the switch from coal to gas results in a fall in the output of all pollutants except solid radioactive waste. For SIM2, the switch from road/air to rail, the results are less clear-cut; emissions of six pollutants decline, but six increase.

8.2.2 Attributing resource use and emissions to final demand deliveries

Input–output methods can be used to account for resource use and/or pollution generation in terms of deliveries to final demand. Consider the case of use of the oil resource first. For the three-industry case, using equation 8.8 with equation 8.7 gives

$$O_1 = r_1X_1 = r_1l_{11}Y_1 + r_1l_{12}Y_2 + r_1l_{13}Y_3$$

$$O_2 = r_2X_2 = r_2l_{21}Y_1 + r_2l_{22}Y_2 + r_2l_{23}Y_3$$

$$O_3 = r_3X_3 = r_3l_{31}Y_1 + r_3l_{32}Y_2 + r_3l_{33}Y_3$$

and adding vertically gives

$$\begin{aligned} O_1 + O_2 + O_3 = {} & (r_1l_{11} + r_2l_{21} + r_3l_{31})Y_1 \\ & + (r_1l_{12} + r_2l_{22} + r_3l_{32})Y_2 \\ & + (r_1l_{13} + r_2l_{23} + r_3l_{33})Y_3 \end{aligned}$$

which can be written as

$$O_1 + O_2 + O_3 = i_1Y_1 + i_2Y_2 + i_3Y_3 \tag{8.10}$$

where $i_1 = r_1l_{11} + r_2l_{21} + r_3l_{31}$ etc. The left-hand side of equation 8.10 is total oil use. The right-hand side allocates that total as between final demand deliveries via the coefficients i. These coefficients give the oil intensities of final demand deliveries, oil use per unit, taking account of direct and indirect use. The coefficient i_1, for example, is the amount of oil use attributable to the delivery to final demand of one unit of agricultural output, when account is taken both of the direct use of oil in agriculture and of its indirect use via the use of inputs of manufacturing and services, the production of which uses oil inputs.

For the data on oil use given above with the data of Table 8.1, the oil intensities are

Agriculture	Manufacturing	Services
0.1525	0.2467	0.1617

which with final demand deliveries of

Agriculture	Manufacturing	Services
600	1500	300

gives total oil use, 510 PJ, allocated across final demand deliveries as

Agriculture	Manufacturing	Services
91.50	370.05	48.51

Note that as compared with the industry uses of oil from which the r_i were calculated, these numbers have more oil use attributed to agriculture and less to manufacturing and services. This reflects the fact that producing agricultural output uses oil indirectly when it uses inputs from manufacturing and services.

In matrix algebra, which would be the basis for doing the calculations where the number of sectors is realistically large, n, the foregoing is

$$O = \mathbf{RX} = \mathbf{RLY} = \mathbf{iY} \tag{8.11}$$

to define the intensities, where

O is total resource use (a scalar)
$\mathbf{R}$ is a $1 \times n$ vector of industry resource input coefficients
$\mathbf{i}$ is a $1 \times n$ vector of resource intensities for final demand deliveries

and $\mathbf{X}$, $\mathbf{L}$ and $\mathbf{Y}$ are as previously defined. The resource uses attributable to final demand deliveries can be calculated as

$$\mathbf{O} = \mathbf{R^*Y} \tag{8.11$'$}$$

where

$\mathbf{O}$ is an $n \times 1$ vector of resource use levels
$\mathbf{R^*}$ is an $n \times n$ matrix with the elements of $\mathbf{R}$ along the diagonal and 0s elsewhere.

With suitable changes of notation, all of this applies equally to calculation concerning waste emissions. Where there are several, m, resources, or types of emissions, being considered, the vector $\mathbf{R}$ in 8.11 becomes an $m \times n$ matrix, and with suitable dimensional adjustments elsewhere, the above carries through, and all the intensities for final demand deliveries can be calculated in one operation.

In the case of CO_2 emissions arising in fossil fuel combustion, it is not necessary to know the emissions levels for each industry, as these can be calculated using data on the fossil fuel inputs to each industry and a standard set of coefficients which give the amount of CO_2 released per unit of a particular fossil fuel burned:[7]

	Tonnes CO_2 per PJ
Natural gas	54 900
Oil	73 200
Black coal	104 100
Brown coal	112 700

In this case, fossil fuel intensities can be converted to CO_2 intensities by using these coefficients and aggregating across the fuels. Table 8.3 gives results so obtained (Common and Salma, 1992a) for Australia for 1986/7 for CO_2 intensities and levels for deliveries to final demand: the figures in parentheses are rankings. A CO_2 intensity is the quantity of CO_2 emitted per unit delivery to final demand. In the first column of Table 8.3 CO_2 units are thousands of tonnes, and final demand delivery units are millions of Australian dollars. The first point to note

[7] Actually slightly different coefficient sets can be found in different sources: for an examination of the sensitivity of results to such variations, see Common and Salma (1992a).

Table 8.3 CO_2 intensities and levels for final demand deliveries, Australia 1986/7

Sector	CO_2[a]	CO_2[b]	Percentage of total
Agriculture, forestry, fishing, hunting	1.8007(6)	13.836(8)	4.74
Mining	0.9854(11)	9.953(12)	3.41
Meat and milk products	1.0368(10)	8.515(13)	2.92
Food products	1.5325(8)	11.540(10)	4.00
Beverages and tobacco	0.9213(12)	3.399(20)	1.17
Textiles, clothing and footwear	0.5561(24)	3.062(21)	1.05
Wood, wood products, furniture	0.8771(14)	2.034(23)	0.70
Paper, products, printing, publishing	0.8707(15)	1.390(24)	0.48
Chemicals	1.2385(9)	2.579(22)	0.88
Petroleum and coal products	10.7272(2)	37.788(2)	12.95
Non-metallic mineral products	2.1980(5)	0.357(26)	0.12
Basic metals, products	4.4977(4)	20.25(4)	6.94
Fabricated metal products	1.7055(7)	3.484(19)	1.19
Transport equipment	0.7406(20)	4.706(17)	1.61
Machinery and equipment	0.8834(13)	5.296(16)	1.82
Miscellaneous manufacturing	0.7727(18)	1.012(25)	0.35
Electricity	15.2449(1)	43.747(1)	14.99
Gas	9.9663(3)	4.675(18)	1.60
Water	0.6680(22)	0.205(27)	0.07
Construction	0.7567(19)	28.111(3)	9.64
Wholesale and retail, repairs	0.4978(25)	18.225(5)	6.25
Transport, storage, communication	0.8157(17)	13.386(9)	4.58
Finance, property, business services	0.6242(23)	5.719(14)	1.96
Residential property	0.1992(27)	5.504(15)	1.89
Public administration, defence	0.8409(16)	14.352(7)	4.92
Community services	0.4437(26)	17.802(6)	6.10
Recreational, personal services	0.7205(21)	10.830(11)	3.71
Total		291.756	100.00

[a] tonnes $\times 10^3/(\$A \times 10^6)$
[b] tonnes $\times 10^6$
Source: Adapted from Common and Salma (1992a)

here is that deliveries of the output of the agriculture, forestry and fishing sector to final demand are relatively CO_2 and fossil-fuel-intensive, ranking sixth, and ahead of several manufacturing sectors. This counter-intuitive result arises because the sector is a large indirect user of fossil fuels, with, particularly, large inputs of fertiliser, the production of which is fossil-fuel-intensive. It means that expansion of Australia's agricultural industry would, per unit, increase Australian CO_2 emissions by more than expansion of several manufacturing outputs. A second point worthy of noting explicitly is that while service sectors – such as wholesale and retail, repairs, or public administration, defence – rank low by intensity, they climb well up the ranking according to emissions levels, on account of their large size. The third point to be made here concerns electricity. It is frequently stated that for Australia – and the case is much the same in most other industrialised economies – electricity generation accounts for approximately 45% of CO_2 emissions associated with fossil fuel combustion. In Table 8.3, electricity accounts for only some 15% of total emissions. It is true that 45% of emissions are through the stacks of power stations. However, much of the electricity so generated is used as input to other productive activities, rather than consumed by households. The accounting in Table 8.3 attributes the emissions associated with electricity as an intermediate commodity to the sectors that use electricity in that way, and the CO_2 total for electricity relates solely to its use by final consumers. For many purposes, accounting for emissions in terms of final demand deliveries is more useful than accounting in terms of the location of fossil fuel combustion. It aligns, as the next section will show, with the impact of carbon taxation on relative prices.

Box 8.2 looks at some recent work using input–output modelling to attribute CO_2 emissions to UK households.

Box 8.2 Attributing CO_2 emissions to UK households

Druckman and Jackson (2009) reports the results of an input–output modelling exercise in examining trends in the CO_2 emissions for which UK households are responsible. Here we report some of those results in a slightly different way, using figures kindly supplied by Angela Druckman. In contrast to the results presented in Table 8.3, those here relate to deliveries to the household consumption category of final demand, not to all deliveries to final demand.

Most exercises of this kind assume that imports are produced using exactly the same technology as domestic production. Druckman and Jackson partially avoid this assumption. In their analysis the rest of the world is represented as 12 regional economies which supply the UK with imports. Each of these economies is assumed to have the same **A** matrix as the UK. Corresponding to our **R** matrix, Druckman and Jackson have a vector of CO_2 coefficients, which vector does differ across economies so as to reflect their differing CO_2 intensities.

Figure 8.1 shows some trends to 2004 using index numbers where the base year is 1992. The total of CO_2 emissions attributable to UK households, including those arising in the production of UK imports, rises by about 20% over the period. The CO_2 intensity of UK household expenditure, CO_2 per £, falls by about 20%. What we called 'indirect use' Druckman and Jackson call 'embedded' CO_2. Figure 8.1 shows the percentage of the CO_2 embedded in UK household consumption which is due to imports increasing over 1992–2004.

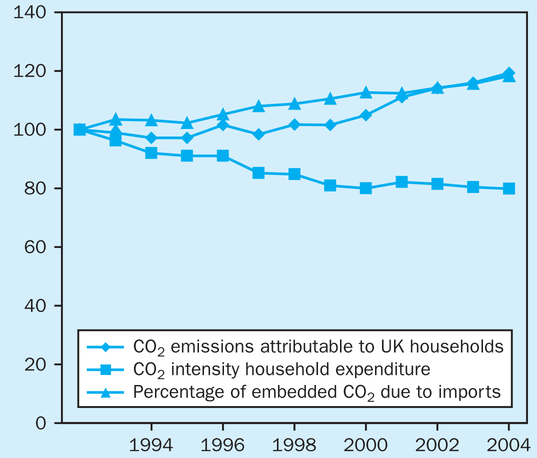


Figure 8.1 Some trends 1992–2004
Source: Druckman and Jackson (2009)

Table 8.4 gives for each year total emissions attributable to UK households and the percentage breakdowns of that total across Embedded, Personal Vehicle Use (running cars etc), aircraft Flights, and Direct which is fuel purchases for domestic heating, lighting, etc. The figures in the first column are the quantities corresponding to the index numbers in Figure 8.1. Over 1992 to 2004 the shares of the total accounted for by Direct and Vehicle Use declined, while those for Embedded and Flights increased.

Table 8.4 CO_2 emissions attributable to UK households 1992–2004

	Total Tonnes $CO_2 \times 10^6$	Percentage			
		Embedded	Vehicle Use	Flights	Direct
1992	467.2	49.3	13.2	2.9	34.6
1993	462.1	50.0	13.4	3.1	33.5
1994	454.1	50.2	13.3	3.2	33.2
1995	456.8	51.8	13.0	3.4	31.8
1996	475.0	50.2	13.2	3.3	33.3
1997	460.0	51.0	13.9	3.6	31.6
1998	475.4	51.2	13.3	3.9	31.6
1999	474.8	51.3	13.7	4.3	30.7
2000	490.2	51.3	13.1	4.8	30.9
2001	518.8	52.2	12.5	4.6	30.6
2002	533.0	53.7	12.6	4.6	29.1
2003	542.1	54.0	12.2	4.6	29.2
2004	557.4	54.5	12.0	4.8	28.7

Source: Druckman and Jackson (2009)

Box 8.2 *continued*

Finally here we report in Table 8.5 some results relating to the attribution of CO_2 emissions to different types of household, known as 'Supergroups'. These results are obtained by aligning the emissions attributable to final demand categories with expenditure patterns for these Supergroups. The names of the Supergroups give a reasonable idea of what sort of households they comprise: for details see Druckman and Jackson (2009) and the references provided there. In Table 8.5 the Supergroups are ordered by the average household level of disposable income.

The first two columns of figures give average tonnes of attributable emissions per household and per capita – average household size varies across Supergroups. The figures in brackets are the rank. Clearly, it does not make much difference to the ranking whether it is done per household or per capita, and the ranking by emissions per household is exactly the same as the ranking by income per household. Whether we look at per household or per capita, 'Prospering suburbs' members are responsible for about 60% more CO_2 emissions than 'Constrained by circumstances' members.

The four rightmost columns in Table 8.5 give the breakdowns across the categories as distinguished in Table 8.4. The differences in the pattern across Supergroups are pretty much what would be expected – most notably, the better-off do more flying and using personal vehicles, while the worst-off spend a larger share of their budgets on direct energy use and hence have the highest Direct responsibility for CO_2 emissions.

Druckman and Jackson (2009) are careful to spell out some of the problems with the data that they use, and note that their work is 'by no means the last word in unraveling the complex mixture of factors that drive modern consumption patterns' and, hence, CO_2 emissions. Francis (2004) conducted a similar, but more limited exercise, looking at all greenhouse gas emissions attributable to UK household consumption in 2001. Francis calculated the total of such to be 612.4 million tonnes of CO_2 equivalent, to be compared with Druckman and Jackson's 518.8 million tonnes of CO_2 itself. Of the total, Francis attributed 158.8 million tonnes to direct use and 456.6 to indirect use, corresponding to embedded for the Druckman and Jackson exercise.

Table 8.5 CO_2 emissions attributable to UK Supergroups 2004

	Tonnes CO_2		Percentage			
	Per household	Per capita	Embedded	Flights	Vehicle Use	Direct
1. Prospering suburbs	26.5 (1)	10.4 (1)	54.0	5.5	12.6	27.9
2. Countryside	24.9 (2)	10.2 (2)	53.8	5.4	12.4	28.4
3. Typical traits	22.4 (3)	9.2 (3)	55.1	5.0	12.3	27.6
4. City living	18.7 (5)	8.3 (4)	56.1	4.3	11.7	27.9
5. Blue collar	19.5 (4)	8.0 (5)	54.0	4.5	12.0	29.6
6. Multicultural	18.2 (6)	7.7 (6)	55.6	4.0	11.7	28.8
7. Constrained by circumstances	16.1 (7)	7.4 (7)	54.2	3.9	11.1	30.8
UK mean	21.5	9.0	54.5	4.8	12.0	28.7

Source: Druckman and Jackson (2009)

8.2.3 Analysing the effects of technical change

In Box 8.1 we reported on the results from a study that used input–output methods to consider whether the technological improvements mooted in the Brundtland Report were sufficient to make sustainable development feasible. In this sub-section we explain how input–output methods can be used to analyse the effects of technical change.

8.2.3.1 Direct energy conservation

In the three-sector hypothetical example looked at previously here, for final demand deliveries

Agriculture	Manufacturing	Services
600	1500	300

the gross sectoral output levels were:

Agriculture	Manufacturing	Services
1000	2000	600

With the oil use coefficients

$r_1 = 0.05$ for agriculture

$r_2 = 0.2$ for manufacturing

$r_3 = 0.1$ for services

total oil use by the economy is 50 + 400 + 60 = 510PJ.

Now, suppose that there is the possibility of a technological innovation in manufacturing that would cut its oil use by 25%, so that r_2 would become 0.15. If such were adopted, the economy's total oil use would be $[(0.05 \times 1000) + (0.15 \times 2000) + (0.1 \times 600)] = 50 + 300 + 60 = 410$PJ, a reduction of 100PJ.

8.2.3.2 Indirect energy conservation

Consider now an innovation in the use of manufacturing output in the production of agricultural output which can be represented as a reduction in the coefficient a_{21} from 0.35 to 0.25. In that case the matrix **A** would be

$$\mathbf{A} = \begin{bmatrix} 0.0000 & 0.2000 & 0.0000 \\ 0.2500 & 0.0000 & 0.2500 \\ 0.1000 & 0.1000 & 0.0000 \end{bmatrix}$$

so that

$$\mathbf{L} = (\mathbf{I} - \mathbf{A})^{-1} = \begin{bmatrix} 1.0598 & 0.2174 & 0.0543 \\ 0.2989 & 1.0870 & 0.2717 \\ 0.1359 & 0.1304 & 1.0326 \end{bmatrix}$$

and for the original deliveries to final demand, gross output levels are:

$$\begin{bmatrix} \mathbf{X}_1 \\ \mathbf{X}_2 \\ \mathbf{X}_3 \end{bmatrix} = \mathbf{LY} = \begin{bmatrix} 1.0598 & 0.2174 & 0.0543 \\ 0.2989 & 1.0870 & 0.2717 \\ 0.1359 & 0.1304 & 1.0326 \end{bmatrix} \begin{bmatrix} 600 \\ 1500 \\ 300 \end{bmatrix}$$

$$= \begin{bmatrix} 978.2609 \\ 1891.3043 \\ 586.9565 \end{bmatrix}$$

With the original oil use coefficients, total oil use in the economy is now $[(0.05 \times 978.2609) + (0.2 \times 1891.3043) + (0.1 \times 586.9565)] = 485.8696$PJ. This is a reduction of 24.1304PJ on the original level.

For the economy as a whole, energy conservation can be achieved indirectly, by reducing the use in production of things the production of which uses energy.

8.2.3.3 Combining direct and indirect energy conservation

Now suppose that both energy conservation in the production of manufacturing output and the new agricultural technology, using less manufacturing output per unit agricultural output, were to be adopted. To figure the effect in this case, we use the energy input coefficients from 8.2.3.1 with the gross output levels that we found in 8.2.3.2. In this case, we find total energy use to be 391.3044PJ, for a reduction of 118.6956PJ. This is, of course, a bigger reduction than in 8.2.3.1 or 8.2.3.2, but it is smaller than the sum of those reductions. This is because in this joint case, the per unit reduction in energy use in the production of manufacturing output is being applied to a smaller gross output level.

8.2.3.4 Australian CO_2 emissions analysis

The work from which Table 8.3 is taken also conducted some simulation experiments exploring the potential for alternative routes to the abatement of CO_2 emissions in Australia. Some examples are as follows. Cutting the final demand for electricity by 10% would reduce total emissions by 1.5%. Cutting the final demand for the output of the construction industry by 10% would reduce total emissions by 1.0%. This result arises because, when indirect use is taken account of, construction is a relatively CO_2-intensive sector, and it is a large sector. If in the matrix of inter-industry coefficients, **A**, those for electricity inputs are all cut by 10%, then for the original set of final demands, total CO_2 emissions reduce by 4.4%. If, instead, the coefficients for basic metal inputs to all industries are cut by 10%, there is a 1.4% reduction in total emissions. Given that the basic metal industry is relatively energy- and CO_2-intensive, conserving on inputs of its commodity is energy-conserving and CO_2-abating. Materials conserving technical change is energy-conserving, and CO_2-abating, because the extraction and processing of materials uses energy, which is currently predominantly based on fossil fuel use.

8.3 Costs and prices

In the preceding sections inputs and outputs were expressed in constant-value terms for economic flows and in physical units for environmental extractions (resource inputs) and insertions (waste emissions). The accounting and analysis were concerned with the 'real' side of the economy, and with questions such as 'if final demand changes, what will happen to emissions?' However, many of the most interesting and controversial issues in environmental economics involve questions of costs and prices. For instance, how would a 'carbon tax' affect the prices facing households, and hence the cost of living?

8.3.1 The dual of the input–output model

These questions can be explored using the dual of the input–output model system outlined above. Analogous to equation 8.1 based on the rows of the transactions table, we can write for the columns of that table

$$X_j \equiv \Sigma_i X_{ij} + M_j + W_j + OVA_j \quad j = 1, \ldots, n \tag{8.12}$$

that is, the value of output of sector j covers the cost of purchases from other sectors $\Sigma_i X_{ij}$, plus the cost of imports used in the production of product M_j, plus labour costs, W_j, plus other value added, OVA_j, which includes profit and is essentially the balancing item in the accounting identity. To simplify the exposition, we aggregate imports, labour costs and other value added, so that

$$X_j \equiv \Sigma_i X_{ij} + V_j, \quad j = 1, \ldots, n \tag{8.13}$$

where V_j is primary input cost. We now assume as before that intermediate inputs are a fixed proportion of industry output, as in equation 8.2. Substituting in equation 8.13 this gives

$$X_j = \Sigma_i a_{ij} X_j + V_j, \quad j = 1, \ldots, n \tag{8.14}$$

Now, the inter-industry flows in the transactions table are expenditure flows, that is price times quantity. When we use the data to consider questions about the 'real' side of the economy, we are dealing with commodities where quantities are measured in units which are 'millions of dollars worth'. Such quantities have, in the accounts, prices which are unity. With P_j for the price of the jth commodity, equation 8.14 can then be written

$$P_j X_j = \Sigma_i a_{ij} P_j X_j + V_j, \quad j = 1, \ldots, n$$

and dividing by X_j gives

$$P_j = \Sigma_i a_{ij} P_j + V_j / X_j, \quad j = 1, \ldots, n$$

or

$$P_j = \Sigma_i a_{ij} P_j + v_j, \quad j = 1, \ldots, n \tag{8.15}$$

where v_j is primary input cost per unit output.

In matrix algebra equation 8.15 is

$$\mathbf{P} = \mathbf{A}'\mathbf{P} + \mathbf{v} \tag{8.16}$$

where $\mathbf{P}$ is an $n \times 1$ vector of prices, $\mathbf{A}'$ is the transpose of the $n \times n$ matrix of input–output coefficients, $\mathbf{A}$, and $\mathbf{v}$ is an $n \times 1$ vector of primary input cost coefficients.

From Equation 8.16

$$\mathbf{P} - \mathbf{A}'\mathbf{P} = \mathbf{v}$$

and with $\mathbf{I}$ as the identity matrix

$$(\mathbf{I} - \mathbf{A}')\mathbf{P} = \mathbf{v}$$

so that

$$\mathbf{P} = (\mathbf{I} - \mathbf{A}')^{-1}\mathbf{v}$$

This last result can be written more usefully as

$$\mathbf{P}' = \mathbf{v}'(\mathbf{I} - \mathbf{A})^{-1} = \mathbf{v}'\mathbf{L} \tag{8.17}$$

where $\mathbf{P}'$ is a $1 \times n$ vector of prices (the transpose of $\mathbf{P}$), $\mathbf{v}'$ is a $1 \times n$ vector of primary input cost coefficients (the transpose of $\mathbf{v}$) and $\mathbf{L}$ is the $n \times n$ Leontief inverse matrix.

According to equation 8.17, commodity prices can be calculated using the Leontief inverse and the primary input cost coefficients. This can be illustrated using data from the transactions table given in Table 8.1. The primary input cost coefficients are:

Agriculture	Manufacturing	Services
0.55	0.70	0.75

Using these in equation 8.17 with the Leontief inverse

$$\mathbf{L} = \begin{bmatrix} 1.0833 & 0.222 & 0.0556 \\ 0.4167 & 1.1111 & 0.2778 \\ 0.1500 & 0.1333 & 1.0333 \end{bmatrix}$$

gives $P_1 = 1.00$, $P_2 = 1.00$ and $P_3 = 1.00$. For $n = 3$, equation 8.14 is

$$X_1 = X_{11} + X_{21} + X_{31} + V_1$$

$$X_2 = X_{12} + X_{22} + X_{32} + V_2$$

$$X_3 = X_{13} + X_{23} + X_{33} + V_3$$

and following the steps above leads to equations 8.15 as

$$P_1 = a_{11}P_1 + a_{21}P_1 + a_{31}P_1 + v_1$$

$$P_2 = a_{12}P_2 + a_{22}P_2 + a_{32}P_2 + v_2$$

$$P_3 = a_{13}P_3 + a_{23}P_3 + a_{33}P_3 + v_3$$

where on substituting for the a and v coefficients, it can readily be confirmed that $P_1 = P_2 = P_3 = 1$ is the solution.

Given that we already knew that all prices are unity in an input–output transactions table, equation 8.17 does not appear very useful. In looking at the 'real' side of things, using the Leontief inverse calculated from the transactions table with the final demand given in the transactions table in equation 8.6 would simply give the gross outputs reported in the transactions table. The usefulness of computing the Leontief inverse from the transactions table was in considering the implications for gross outputs, and flows to and from the environment, of different levels and patterns of final demand. Here, the point is that equation 8.17 can be used to consider the commodity price implications of v coefficients other than those derived from the transactions table.

Suppose that for the hypothetical economy to which Table 8.1 refers, carbon taxation were under consideration. In the preceding section we gave data on the use of oil by each of the three sectors, and noted that the use of 1 PJ of oil meant the emission of 73.2×10^3 tonnes of CO_2. This means that the CO_2 emissions arising in each sector are, in kilotonnes = 10^3 tonnes:

Agriculture	Manufacturing	Services
3660	29 280	4392

Suppose that the rate of carbon taxation under consideration is \$20 per tonne. From equation 8.17 the change in prices for a change in the v coefficients is

$$\Delta\mathbf{P}' = \Delta\mathbf{v}'\mathbf{L} \qquad (8.18)$$

where $\Delta\mathbf{v}'$ is the transposed vector of changes in the primary input cost coefficients and $\Delta\mathbf{P}'$ is the transposed vector of consequent price changes. For the postulated rate of carbon taxation, using the figures above for emissions and the data from Table 8.1 gives

$$\Delta v_1 = 0.0682, \quad \Delta v_2 = 0.2265, \quad \Delta v_3 = 0.1277$$

for which equation 8.18, with $\mathbf{L}$ as given above, yields

$$\Delta P_1 = 0.1874, \quad \Delta P_2 = 0.2838, \quad \Delta P_3 = 0.1987$$

Given that prior to the imposition of the carbon tax all the prices were unity, these are proportionate price increases; that is, the price of the commodity which is the output of the agricultural sector would increase by 18.74%. Note, for example, that whereas the manufacturing sector uses four times as much oil per unit gross output, and hence emits four times as much CO_2, as the agriculture sector, the ratio of price increases in manufacturing relative to agriculture, 1.5, is smaller than four. Using input–output analysis picks up the implications for pricing of the fact that the agriculture sector uses oil, so that delivering its output to final demand is responsible for CO_2 emissions, indirectly as well as directly.

It should also be noted that this analysis involves the assumption that the input–output coefficients, the a_{ij}, and the coefficients for oil inputs do not change in response to the imposition of a carbon tax. It involves the assumption, that is, that making oil inputs more expensive does not induce any substitution responses on the part of producers. In so far as any such responses would involve using less oil per unit output, and less of relatively oil-intensive commodities as intermediate inputs, they would reduce the price increases consequent on the introduction of the carbon tax. The input–output results can, that is, be regarded as setting upper bounds to the price increases that would actually occur. In so far as substitution responses take time to implement, it would be expected that those upper bounds would approximate the short-run impacts, rather than the long-run impacts.

Box 8.3 Input–output analysis of rebound in Spanish water supply

Llop (2008) uses input–output analysis to examine rebound, and measures to offset it, in the context of the use of water as an input to production in Spain. The study exemplifies how input–output modelling's limitations in regard to lacking behavioural responses can, with a little ingenuity, be partially overcome in policy analysis.

Llop uses an 18 × 18 **A** matrix for Spain for 2000, where industry 18 is the water supply industry, and uses the pricing model discussed here to calculate the change in the prices of each of the 18 commodities in each of three scenarios:

1. The *a* coefficients in row 18 are reduced by 20%, and those in column 18 are also increased by 20%. The former corresponds to an across the board improvement in the efficiency with which water is used in production, and the latter is equivalent to an increase in the efficiency with which water is supplied.
2. The imposition of a 40% tax on the price that industries pay for their water.
3. The combination of scenarios 1 and 2.

With $j = 1, \ldots 18$, P_{j0} is the price of the jth commodity initially and P_{j1} is the price after the imposition of the scenario change, and similarly for X_{j0} and X_{j1}. Let k be the ratio, the same across all sectors, by which expenditure changes when price changes, so that

$$P_{j1}X_{j1} = \mathrm{k}P_{j0}X_{j0}$$

and with $P_{j0} = 1$ for all j, this means

$$X_{j1} = \mathrm{k}(X_{j0}/P_{j1}).$$

gives the quantity demanded by industry j following a price change.

Table 8.6 Changes in total industrial water use

	Scenario 1	Scenario 2	Scenario 3
Water Use change %			
Expenditure constant, k = 1	20.08	–28.65	–14.30
Expenditure down 10%, k = 0.9	8.07	–35.79	–22.87
Expenditure up 10%, k = 1.1	32.09	–1.52	–5.73

Source: Llop (2008)

The rows of Table 8.6 give the percentage changes in total water use by all industries under each of the three scenarios for three different assumptions about k. With unitary elasticity of demand, k is equal to 1, and expenditure is constant when price changes. The case k equal to 0.9 corresponds approximately to an elasticity of demand of 0.9, and k equal to 1.1 to an elasticity of 1.1. The results show, first, that the change in total industrial water use is very sensitive to the elasticity of demand. Second, the Scenario 1 results show that, for the demand elasticities considered, improvements in water efficiency do lead to greater use of water – there is a rebound effect. Third, the Scenario 3 results show that rebound can be more than offset by the introduction of a tax on water use by industry.

Given software for doing the matrix calculations, it would not take much time to experiment with different rates of tax, so as to find, under each of the elasticity assumptions, the tax rate that would exactly offset the rebound effect, rather than overcompensate as in Table 8.6. Equally, a wider range of possible values for k could be explored.

8.3.2 The regressivity of carbon taxation

Table 8.7 gives the results of calculations essentially the same as those described above using the same input–output data for Australia as were used for the results given in Table 8.3. The calculations differ in so far as there are several fuels, rather than just oil, used, and in so far as the Australian input–output data involve a distinction between the prices received by sellers and those paid by buyers, which reflects indirect taxation and the way in which the accounts treat distribution margins. It is because of the latter that the rankings in Table 8.7, shown in parentheses, do not exactly match those shown in Table 8.3 for the CO_2 intensities of deliveries to final demand.

It is widely believed that carbon taxation would be regressive in its impact, would hurt the poor more than the rich. Input–output analysis of the impact on commodity prices can provide one input to a quantitative analysis of this question. The other necessary input is data on the expenditure patterns of households at different positions in the income distribution

Table 8.7 Price increases due to a carbon tax of A$20 per tonne

Sector	Percentage price increase	
Agriculture, forestry, fishing, hunting	1.77	(9)
Mining	1.69	(12)
Meat and milk products	1.77	(9)
Food products	1.46	(16)
Beverages and tobacco	0.84	(24)
Textiles, clothing and footwear	0.95	(21)
Wood, wood products, furniture	1.31	(15)
Paper, products, printing, publishing	1.12	(20)
Chemicals	1.56	(16)
Petroleum and coal products	9.97	(4)
Non-metallic mineral products	1.89	(8)
Basic metals, products	9.00	(5)
Fabricated metal products	2.76	(6)
Transport equipment	0.82	(23)
Machinery and equipment	0.71	(26)
Miscellaneous manufacturing	0.89	(23)
Electricity	31.33	(1)
Gas	21.41	(2)
Water	1.34	(18)
Construction	1.60	(13)
Wholesale and retail, repairs	10.14	(3)
Transport, storage, communication	2.28	(7)
Finance, property, business services	1.21	(19)
Residential property	0.42	(27)
Public administration, defence	1.73	(11)
Community services	0.93	(21)
Recreational, personal services	1.62	(13)

Source: Adapted from Common and Salma (1992b)

Table 8.8 CPI impacts of carbon taxation

Decile	Accounting for direct and indirect impacts %	Accounting for only direct impacts %
1	2.89	1.53
2	3.00 H	1.66 H
3	2.97	1.60
4	2.85	1.44
5	2.88	1.45
6	2.77	1.35
7	2.80	1.31
8	2.77	1.28
9	2.67	1.16
10	2.62 L	1.10 L
All households	2.79	1.31
H/L ratio	1.15	1.51

Source: Adapted from Common and Salma (1992b)

which is, or can be made, compatible with the input–output data in terms of its commodity classification. Where such data are available, the change in the cost of living for a household is given by

$$\Delta \text{CPI}_h = \Sigma_j \beta_{hj} \Delta P_j, \quad h = 1, \ldots, m \tag{8.19}$$

where CPI stands for consumer price index, h indexes households, and β_{hj} is the budget share of commodity j for the hth household. Table 8.8 gives results for Australia, using the price changes from Table 8.7 here with data on Australian household expenditure patterns by expenditure decile.[8] In Table 8.8 H identifies the highest CPI effect, L the lowest. The presumption that carbon taxation would be regressive in its impact comes from the observation, generally valid for industrial economies, that lower-income groups spend a larger proportion of their income on fuel than upper-income groups. The third column in Table 8.8 shows the CPI impacts when j in equation 8.19 indexes only the fuel commodities: electricity, gas, and petroleum and coal products. The second column shows the CPI impact when j indexes all 27 commodities, picking up the indirect as well as the direct price effects of carbon taxation. There is by the H/L ratio a regressive impact in both columns, but it is smaller in the second column. Just looking at direct fuel purchases overstates the regressive impact of carbon taxation (and, equivalently, of higher fuel prices). Carbon dioxide taxation affects all commodity prices, roughly, in proportion to their carbon dioxide intensity. Thus, while the poor spend proportionately more on direct fuels purchases, the rich spend more on things, such as overseas travel, in which fuels are, directly and indirectly, used as inputs, and this reduces regressivity.

Two points need to be kept in mind when considering results such as those shown in Table 8.8. The first is that using equation 8.19 involves the assumption that household expenditure patterns do not change in response to the changed relative prices induced by carbon taxation. The assumption of fixed budget shares for households is directly analogous to the assumption of fixed input–output coefficients in production, and has similar implications for the results of the analysis. Given changed relative prices

[8] The household expenditure data is for 1984, which was at the time that the study was done the most recent such data available by decile. For results using more recent household expenditure data, by quintile, see Common and Salma (1992b), which also gives a more detailed account of data and methods.

following the introduction of carbon taxation, households would be expected to substitute commodities for which price had risen less for those for which it had risen more. Such behaviour would reduce the CPI impact of carbon taxation: results based on equation 8.19 would, for each decile, represent an upper limit. The second point is that the impacts on households arising from carbon taxation would not be confined to the expenditure side. There would also be income effects. As noted above, producers would also be expected to make substitution responses, which would have implications for employment opportunities and incomes. Also, carbon taxation would give rise to government revenue which could be used in a variety of ways, including increased welfare payments and/or lower income taxation for low-income households, for example. If substitution responses, and discretion in the use of the tax receipts, are to be allowed for, analysing the full implications of the introduction of carbon taxation, and other environmental policies, is beyond the scope of input–output analysis. Such issues can, in principle, be investigated using the methods to be discussed in the next section.

8.4 Computable general equilibrium models

Environmental input–output models are undoubtedly useful for applied work in policy simulation, forecasting and structural analysis. They are transparent and computationally straightforward. However, they are seen by many economists as suffering from several serious deficiencies. Utility- and profit-maximising behaviour play no role in input–output models: there are no demand and supply equations and no capacity constraints. Concern with the rather limited behavioural basis of input–output models has led to a growing interest in applied, or computable, general equilibrium models. We will use the term 'computable general equilibrium' (CGE) models.

CGE models are essentially empirical versions of the Walrasian general equilibrium system and employ the theoretical (neoclassical) assumptions of that system, which, as remarked above, are absent from the input–output system. In general, CGE models cannot be solved algebraically, but thanks to recent increases in computing power, and the development of solution algorithms, they can be solved computationally. These developments have stimulated a rapid growth in applied CGE modelling, particularly on issues related to taxation, trade, structural adjustment and the environment.

8.4.1 An illustrative two-sector model

Here we will use constructed data for an imaginary economy and a simple CGE model of that economy to illustrate the essentials of CGE modelling in relation to environmental problems.[9]

8.4.1.1 The economy

Table 8.9 is the transactions table for a two-sector economy. This economy uses labour and oil as inputs to production, and in Table 8.9 'Other value added' refers to payments to the owners of oil deposits. The units of measurement are $\$ \times 10^6$ everywhere. The only component of final demand is consumption by the households which supply labour and own oil deposits – there is no foreign trade. The implication that households sell costlessly extracted oil to the

Table 8.9 Transactions table for the two-sector economy

	Agriculture	Manufacturing	Consumption	Total output
Agriculture	0	1.3490	3.1615	4.5105
Manufacturing	1.1562	0	3.1615	4.3177
Wages	2.5157	1.4844		
Other value added	0.8386	1.4843		
Total input	4.5105	4.3177		

[9] For a good summary of work on CGE modelling and applications see Greenaway *et al.* (1993).

Table 8.10 Physical data for the two-sector economy

	Agriculture	Manufacturing	Consumption	Total output
Agriculture	0	0.5508	1.2909	1.8417
Manufacturing	0.3687	0	1.0083	1.3770
Labour	2.5157	1.4844		
Oil	0.7217	1.2774		
Emissions	52.8484	93.5057		

producing sectors of the economy keeps what follows as simple as possible, while not affecting the essentials.

The prices of both produced commodities and of both primary inputs are known for the year to which Table 8.9 relates. Since, as we shall see, it is for present purposes only relative prices that matter, we set the price of labour at unity and express all other prices in terms of the 'wage rate' as numeraire. Then, the known prices are:

Agriculture	$P_1 = 2.4490$
Manufacturing	$P_2 = 3.1355$
Labour	$W = 1$
Oil	$P = 1.1620$

With these prices we can convert the transactions table into an input–output table expressed in physical units (Table 8.10). Take it that the units in Table 8.10 are: tonnes $\times 10^6$ for agriculture and manufacturing; person-years for labour and PJs for oil. Table 8.10 has an additional row labelled 'Emissions'. Each PJ of oil used gives rise to 73.2×10^3 tonnes of CO_2, and in Table 8.10 emissions are reported in units of kilotonnes.

8.4.1.2 The model

Suppose now that we want to consider policy in regard to CO_2 emissions for this economy. If we restrict ourselves to the assumptions of input–output modelling, we can, as discussed in the second section of this chapter, consider the implications of alternative final demand scenarios, and/or the implications of changes to the economy's technology, as reflected in the matrix **A**. We could also, as described in Section 8.3, consider the implications for commodity prices of the imposition of a carbon tax at various rates. However, an input–output model would not give a result for the reduction emissions due to the introduction of the tax as there is no representation in it of how producers and consumers respond to the higher commodity prices that the tax causes. Notice also that while an input–output model could give a number for the tax revenue, as tax rate times emissions, that number would be an overestimate as emissions would be the pre-tax level of emissions.

The argument for CGE modelling is that by using the assumptions of general equilibrium theory, we can do more useful policy analysis, where agents respond to the policy intervention. Given such responses in the model, it can yield an estimate of the reduction in total emissions, and for the emissions tax revenue. Essentially, CGE modelling employs four sorts of assumption:

1. market clearing – all markets are in equilibrium;
2. Walras's law – all markets are connected;
3. utility maximisation by households;
4. profit maximisation by firms.

Using assumptions (1) and (2) is a relatively straightforward matter of model specification. Using assumptions (3) and (4) requires additional assumptions and/or data.

In regard to (1), consider for example the commodity markets. The market clearing assumption there is that for each commodity, the amount produced is taken off the market by the sum of all the demands. Here, given intermediate uses of produced commodities, that is

$$X_1 = X_{11} + X_{12} + C_1$$

$$X_2 = X_{21} + X_{22} + C_2$$

In regard to the use of intermediate goods in production, we will make the standard input–output modelling assumption, equation 8.2. In that case, we have

$$X_1 = a_{11}X_1 + a_{12}X_2 + C_1$$
$$X_2 = a_{21}X_1 + a_{22}X_2 + C_2 \quad (8.20)$$

In regard to (2), we know, for example, that for this simple economy

$$Y = W(L_1 + L_2) + P(R_1 + R_2) \quad (8.21)$$

where Y is total household income, W is the wage rate, L_i is labour used in the ith sector, P is the price of oil and R_i is oil used in the ith sector.

Together with demand equations for primary inputs and demand and supply equations for produced commodities, equation 8.21 ties together various markets in the economy.

The derivation of numerical demand and supply equations from the assumptions of utility and profit maximisation is less straightforward. Consider utility maximisation and household commodity demands. We will assume, as is typical in CGE modelling, that there is just one household. The general form of the commodity demand equations is then:

$$C_1 = C_1(Y, P_1, P_2)$$
$$C_2 = C_2(Y, P_1, P_2) \quad (8.22)$$

If it were the case that we had adequate time series data on this economy for C_1, C_2, Y, P_1 and P_2 we could use it to test alternative functional forms for these demand equations, and to estimate the parameters for the preferred functional form. In fact, this 'econometric' approach is generally not adopted in actual CGE modelling exercises, as there are not adequate time series data. The alternative, and widely used, approach is known as 'calibration'. This involves assuming some plausible functional form and setting its parameters so that the resulting equations are numerically consistent with the available data. Very often this 'benchmark' data is for a single year, as is the case with Table 8.9, the associated price data, and Table 8.10.

Let us assume, then, that the utility function for the household in this economy is

$$U = C_1^{\alpha}C_2^{\beta} \quad (8.23)$$

Maximising equation 8.23 subject to the budget constraint

$$Y = P_1C_1 + P_2C_2$$

leads, as shown in Appendix 8.2, to the demand functions:

$$C_1 = [\alpha/(\alpha + \beta)P_1]Y$$
$$C_2 = [\beta/(\alpha + \beta)P_2]Y \quad (8.24)$$

Using the data for Y, P_1, P_2, C_1 and C_2, these can be written as

$$\alpha/(\alpha + \beta) = 0.5$$
$$\beta/(\alpha + \beta) = 0.5 \quad (8.25)$$

which are two equations in two unknowns. Unfortunately, and not untypically, equations 8.25 do not have a unique solution for the values of α and β. The solution to equations 8.25 is $\alpha = \beta$. We need some more information. A typical approach in practice would be to 'import' a value for one of these parameters as estimated with some other data for a different economy. We shall simply impose the plausible value $\alpha = 0.5$.

As regards the other sources of demand for produced commodities, numerical parameterisation is straightforward, given that we are making the standard input–output assumptions about intermediate demands. From Table 8.7 we derive the matrix of, physical, input–output coefficients as:

$$\mathbf{A} = \begin{bmatrix} 0 & 0.4 \\ 0.2 & 0 \end{bmatrix}$$

Now consider the production side of the model. We assume, as is typical in CGE practice, that each sector comprises a single firm, which behaves as a price-taker in its output market and the factor markets. We assume constant returns to scale and Cobb–Douglas production functions, with labour and oil as arguments. While the Cobb–Douglas assumption is adopted here mainly for simplicity, the first is generally used in actual CGE modelling, and has important consequences for model structure. As shown in Appendix 8.2, with constant returns to scale, profits are zero at all output levels and so there is no supply function. It is then necessary to construct the model such that firms produce to meet demand. As shown in Appendix 8.2, given the output level, the assumption of cost minimisation means that equations for factor demands per unit output can be derived. There remains the problem of fixing numerical values for the parameters of the production functions, and hence, the factor demand equations. The situation in regard to the production and factor demand side of the economy is as discussed above for household demand equations. While econometric estimation is possible in principle, it is generally, though not always, precluded by the non-availability of the necessary data. Most usually, numerical parameter values are determined by importing some of them from other sources – or on grounds of 'plausibility' – and determining the remainder by calibration against the benchmark data set, here Table 8.10.

We shall not go into this any further here beyond saying that Box 8.4 lists the equations of our CGE model with numerical parameter values which pass the calibration test – running the model listed there does, as shown in the lower part of the box, reproduce the data of Tables 8.9 and 8.10, and give the relative prices that go with that data. Equations (1) and (2) in the box are the household commodity demands, derived as discussed above. Equations (3) and (4) are the commodity balance equations using the values from the matrix **A** given above. Then we have a pair of simultaneous equations in P_1 and P_2. Given that U_{L1} and U_{R1}, for example, are respectively the use of labour and oil per unit output in sector 1, equations (5) and (6) are the same as the pricing equations used in input–output analysis – see equation 8.15 from the previous section – and go with zero profit in each line of production.

There follow eight equations relating to factor demands. Equations (7), (9), (11) and (13) give the quantity of factor input per unit output, and equations (8), (10), (12) and (14) convert the results to factor demand levels using the corresponding output levels. The form of these equations is derived from cost minimisation, as described in Appendix 8.2. The numerical values appearing in equations (7), (9), (11) and (13) in Box 8.4 go with the following numerical specifications for the production functions:

$$\begin{aligned} X_1 &= L_1^{0.75}R_1^{0.25} \\ X_2 &= L_2^{0.5}R_2^{0.5} \end{aligned} \tag{8.26}$$

Equation (15) gives the total of emissions as the sum of the emissions of CO_2 associated with oil combustion in each sector. Equation (16) is equation 8.21, giving total household income as arising from the sales of labour services and oil to both producing sectors. Finally, equations (17) and (18) say that there is a fixed total amount of each factor, L^* and R^*, available to the economy, and that there is full employment of the available amount, taking the two sectors together. These factor endowments are exogenous variables in this model. There are 18 endogenous variables: W, P, Y, E; and for $i = 1, 2$ U_{Li}, U_{Ri}, C_i, P_i, X_i, L_i and R_i.

8.4.1.3 Model simulations

The solution algorithm is based on the economic idea of price moving to clear excess demand/supply, exploits the fact that we are concerned only with relative prices, and is simplified by using the Walras law.[10] It first takes in the numerical values for the parameters and the given total factor endowments L^* and R^*. In the light of the concern for relative prices, labour is selected as numeraire, and W is set at some fixed value, which, given the discussion of the data above, is 1. Given an assumed, temporary, value for P the next step is to use equations (7), (9), (11) and (13) from Box 8.4 to calculate the unit factor demands. These are used with the solution to the commodity pricing equations, equations (5) and (6) (the nature of which solution was discussed in the previous section of the chapter), to derive commodity prices, and with an assumed, temporary, value for X_1 to find L_1 according to equation (8) and R_1 according to equation (12). L_2 is then calculated as $L^* - L_1$. We can then find X_2 from L_2 and the unit factor demand for labour in manufacturing, and given this value for X_2 the manufacturing demand for oil can be found using equation (14). Given values for L_1, L_2, R_1 and R_2, Y can be calculated according to equation (16), and hence household commodity demands from equations (1) and (2).

At this point, the value of $R_1 + R_2$ is compared with that for R^*. If $R_1 + R_2$ is greater than R^*, the value of P is increased by a small amount, to reduce

[10] The algorithm is an adaptation of that listed in chapter 4 in Dinwiddy and Teale (1988).

Box 8.4 The illustrative CGE model specification and simulation results

Computable general equilibrium model specification

(1) $C_1 = Y/2P_1$
(2) $C_2 = Y/2P_2$
(3) $X_1 = 0.4X_2 + C_1$
(4) $X_2 = 0.2X_1 + C_2$
(5) $P_1 = 0.2P_2 + WU_{L1} + PU_{R1}$
(6) $P_2 = 0.4P_1 + WU_{L2} + PU_{R2}$
(7) $U_{L1} = [3(P/W)]^{0.25}$
(8) $L_1 = U_{L1}X_1$
(9) $U_{L2} = [P/W]^{0.5}$
(10) $L_2 = U_{L2}X_2$
(11) $U_{R1} = [0.33(W/P)]^{0.75}$
(12) $R_1 = U_{R1}X_1$
(13) $U_{R2} = [W/P]^{0.5}$
(14) $R_2 = U_{R2}X_2$
(15) $E = E_1 + E_2 = e_1R_1 + e_2R_2$
(16) $Y = W(L_1 + L_2) + P(R_1 + R_2)$
(17) $L_1 + L_2 = L^*$
(18) $R_1 + R_2 = R^*$

Table 8.11 Computable general equilibrium model results

	Base case A	Base case B	50% emissions reduction	Reduction case as proportion of base case
W	1	5	1	1
P	1.1620	5.7751	2.3990	2.0645
P_1	2.4490	12.2410	3.0472	1.2443
P_2	3.1355	15.6702	4.3166	1.3767
X_1	1.8416	1.8421	1.4640	0.7950
X_2	1.3770	1.3770	1.0341	0.7510
L_1	2.5157	2.5164	2.3983	0.9533
L_2	1.4844	1.4836	1.6017	1.0790
R_1	0.7216	0.7226	0.3332	0.4618
R_2	1.2774	1.2780	0.6677	0.5227
R	2	2	1	0.5000
E_1	52.8484	52.8484	24.3902	0.4618
E_2	93.5057	93.5057	48.8756	0.5227
E	146.3541	146.3541	73.2658	0.5000
Y	6.324	31.615	6.3990	1.0119
C_1	1.2909	1.2912	1.0503	0.8136
C_2	1.0083	1.0087	0.7415	0.7354
U	1.1409	1.1412	0.8825	0.7735

the excess demand for oil, and the calculation described in the previous paragraph repeated. If $R_1 + R_2$ is less than R^*, the value of P is reduced, to reduce the excess supply of oil, and the calculation described in the previous paragraph repeated. These iterations are repeated until a value for P is found for which, to some close approximation, $R_1 + R_2 = R^*$. At this point, the iteration process ceases. We know by virtue of the calculations as described that the oil market, the X_1 market, and the labour market are in equilibrium. And, by virtue of Walras's law, we then know that the remaining market, for X_2, must also be in equilibrium. The computer program which implements the algorithm reports the values of all of the endogenous variables, and stops.

The results for base cases A and B are those that the model produces when L^* is set at 4 and R^* is set at 2, the total labour and oil use in Table 8.10, and when $e_i = 73.2$, as indicated by the data there. The input to the two base case runs of the model differs only in the value given to W, 1 in A and 5 in B. The point of reporting results from these two runs is to illustrate the point that in a CGE model it is only relative prices that matter. Comparing the columns, we see, first, that for B the entries for W, P, P_1 and P_2 are all five times those for A, and, second, that all of the remaining entries, for the 'real' variables, are the same (leaving aside inevitable small differences due to the impact of the rule for stopping iterations in the algorithm). We also see that base case A reproduces the data on prices and quantities that we started with, the model calibrates.

The results in the third column arise when the model is run with base case A input, except that R^* is set at 1, 50% of the total amount of oil used in the original data set. Because emissions are linked to oil use by fixed coefficients, cutting total oil use by 50% will cut emissions by 50%. The results in the fourth column show those in the third as a proportion of those in the first. Looking at the results for the 50% emissions cut in comparison with base case A, we see first that P, the price of oil, increases by more than 100%. The prices of commodities 1 and 2 increase, with P_1 increasing less than P_2. Consistent with this, X_1, the total output of 1, falls by less than X_2. It is not the case that all factor inputs fall. Labour use in the production of commodity 1 goes down, but labour use in 2 goes up – recall that the model is structured so that there is always full employment of the labour available, so that in a two-sector model a reduction in labour input in one sector must be balanced by an increase in the other.

As a result of the reduced availability and hence higher price of oil, both producing sectors use less oil, and produce smaller amounts of emissions. Note, however, that it is not the case that R_1 and R_2, and E_1 and E_2, are reduced by equal amounts, of 50%. Oil use and emissions fall by more than 50% in the production of commodity 1, by less than 50% in the production of commodity 2. This is the efficient loading of total abatement across sources discussed in Chapters 6 and 7, arising because the model mimics competitive firms responding to their relative cost structures.

Household consumption of both commodities falls, with that of the commodity, 1, the price of which rose least, falling least. National income, here the simple sum of household incomes, which is equal to $P_1C_1 + P_2C_2$, increases because the rise in the price of oil more than compensates for the reduced quantity used, and the additional income is more than absorbed by the higher prices for C_1 and C_2. National income as reported in Box 8.4 is not corrected for the change in the general price level, and so misrepresents the welfare change, to the extent in this case of getting the sign for the direction of movement wrong. However, utility as a function of quantities consumed falls. In regard to this, it is important to note that the model has utility dependent only on commodity consumption levels. Emissions are not an argument in the utility function. This is typical of CGE models used for the analysis of the effects of programmes to reduce emissions or improve the environment. To the extent that people do derive benefit from reduced pollution, considering the utility change computed in a CGE model which does not allow for that benefit, means that the reported utility change will be an underestimate. Again, the extent of this may be such that the direction of change is misrepresented – the unrecognised utility gain from reduced emissions could be larger than the utility loss due to reduced commodity consumption.

In this model the reduction in total oil use and emissions has to be simply imposed as action by a *deus ex machina*, and the impact is transmitted through a higher price for oil, which is received directly by the owners of oil deposits, who are the household sector. An actual study of the implications of prospective action on carbon dioxide emissions would be looking not at action by a *deus ex machina* but at some kind of change of policy by government. While it could be argued that a model without a government sector, like that considered above, gives some kind of first-cut feel for the implications of reducing emissions, it is clear that an interesting analysis of government policy requires that the government sector be explicitly represented in the model. Without a government sector, we would have to treat, for example, tradable emissions (or oil use) permits as equivalent to emissions (or oil) taxation, whereas the interesting questions are about how they compare. About the only interesting policy question that the above model could address is a comparison of an efficient policy – permits or tax – with an inefficient policy – making all sources cut in equal proportional amounts. Clearly, it is also the case that to be useful for policy analysis for a trading economy a CGE model would also need a trade sector. Extending the model in these ways makes it complex, and requires more data, or more assumptions about parameter values.

Rather than develop the illustrative model further in these directions, we now look briefly at results from substantive exercises which serve to illustrate the sort of analysis that can be done when trade and government are explicitly represented in a CGE model.[11] Box 8.5 reports some results from an exercise looking at the question of rebound effects in the industrial use of energy in the UK. In the next two sections we look at two GCE modelling exercises dealing with aspects of greenhouse gas abatement. As discussed in Chapter 9, CGE models have been extensively used to analyse policy for the abatement of greenhouse gas emissions, focusing particularly on carbon dioxide emissions.

Box 8.5 CGE modelling of energy rebound in the UK

Allan *et al.* (2007) report results from simulations of their UKENVI model of the UK economy. This model has three classes of agent – households, firms and government. It distinguishes 25 commodities/industries, five of which are energy commodities. The rest of the world is treated as a single economy, ROW. The model is calibrated on a database for 2000, which is largely derived from the 1995 UK input–output table.

Figure 8.2 shows the production structure for each industry. At each level, the two inputs go into a Constant Elasticity of Substitution, CES, production function with output which is the level above. Thus, for example, renewable and non-renewable are inputs to the production of electricity, which is used with non-electricity to produce the composite commodity energy. With X_1 and X_2 as two inputs, and Q as output, a CES production function takes the form:

$$Q = A[\delta X_1^{-\rho} + (1 - \delta)X_2^{-\rho}]^{-1/\rho} \; : \quad A > 0,\ 0 < \delta < 1,\ -1 < \rho < \infty$$

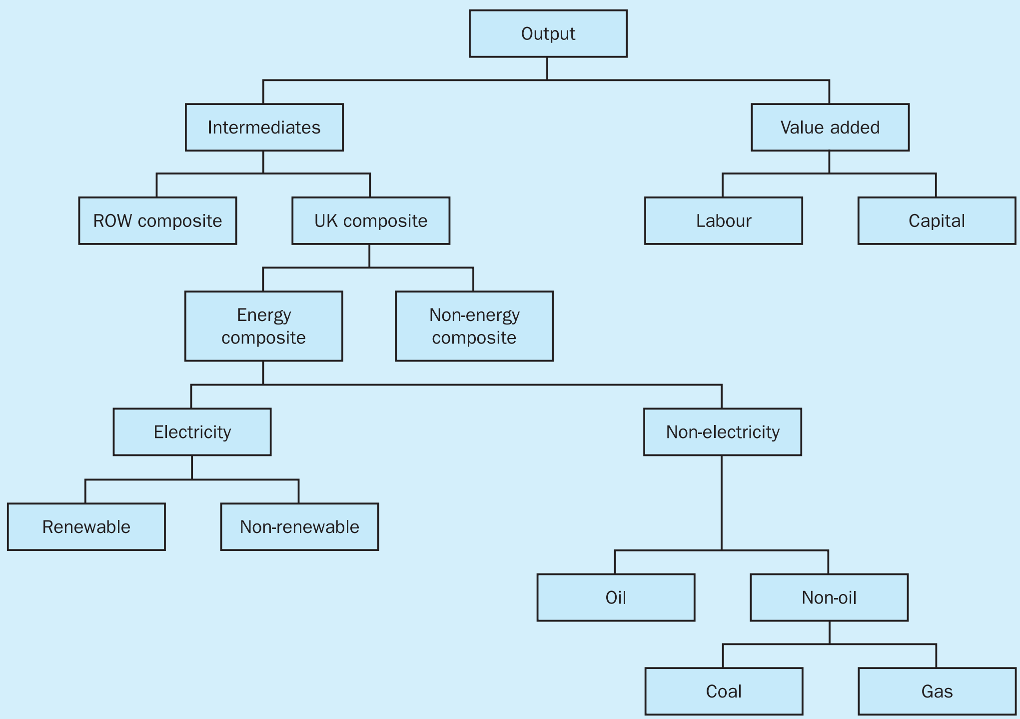


Figure 8.2 Production structure of the UKENVI model

[11] Dinwiddy and Teale (1988) develop, in terms of the algebra and solution algorithms, an illustrative two-industry model to include government expenditure and taxation and foreign trade.

Box 8.5 *continued*

As indicated by the name, a key feature of such a production function is that the elasticity of substitution between the inputs is constant over all levels of the inputs. Its size depends only on the value of the parameter ρ. Using η for the elasticity of substitution, $-1 < \rho < 0$ gives $\eta > 1$, $\rho = 0$ gives $\eta = 1$, $0 < \rho < \infty$ gives $\eta < 1$. The Cobb–Douglas production function is a special case of the CES for $\rho = 0$ and has $\eta = 1$.[12]

In Allan *et al.* (2007) UKENVI is used to investigate rebound in the industrial use of energy in the UK. This relates to the possibility that improvements in energy efficiency are partially, or possibly wholly, offset by consequent increases in the demand for energy by industrial users. The point is that improvements in energy efficiency mean a lower effective price for energy leading to the substitution of energy for other inputs. Also, such improvements may be mean lower total costs of production and an increase in GDP, working to increase energy demand. Allan *et al.* (2007) note the possibility of what they call 'backfire' where the energy efficiency improvement actually leads, eventually, to an increase in the total demand for energy. They argue that rebound/backfire is essentially an empirical matter, and that CGE models are well suited for its investigation given that they explicitly model feedbacks working via maximising behaviour in production and consumption. Recall that, in Box 8.3, we looked at the use of the input–output pricing model to look at rebound in the Spanish water industry.

Allan *et al.* measure rebound as follows. Let ΔE^E be the initial percentage change in energy efficiency and ΔE^M be the percentage change in total energy use after the economy has responded to the initial shock, and let R stand for percentage rebound. Then

$$R = 100 \times \left[1 - \left(\frac{\Delta E^M}{\Delta E^E}\right)\right]$$

and with $\Delta E^E < 0$, four cases can be distinguished:

1. $\Delta E^M < 0$ and greater in absolute value than ΔE^E implies $R < 0$.
2. $\Delta E^M < 0$ and equal in absolute value to ΔE^E implies $R = 0$.
3. $\Delta E^M < 0$ and smaller in absolute value than ΔE^E implies $0 < R < 100$.
4. $\Delta E^M > 0$ implies $R > 100$.

Case 1 would be null in a world where all agents responded rationally to a cut in the effective price of energy, as they do in CGE models. Case 4 is backfire.

In Table 8.12 we reproduce a small sample of the results from Allan *et al.* (2007), where more variables are reported on across more simulations. All of the results in Table 8.12 come from 'long-run' simulations in which sectoral capital inputs change, optimally, in response to the effects of the initial shock. Allan *et al.* also report results for short-run simulations, in which sectoral capital stocks are held at their initial levels. The initial shock is an, exogenous, improvement of 5% in the efficiency with which all industries use energy at the third input level in Figure 8.2. The results under Central case in Table 8.12 are for where this shock is applied for the model with all parameter set at calibrated values. The first three rows show three macro

Table 8.12 Selected simulation results from UKENVI

	% Change from base year					
	Central case	Elasticity of substitution reduced	Constant costs	Government expenditure adjusts	Exogenous labour supply	Real wage resistance
GDP	0.17	0.16	−0.33	0.20	−0.04	0.90
Employment	0.21	0.21	0.03	0.26	0.00	0.95
CPI	−0.27	−0.23	0.17	−0.13	−0.10	−0.68
	Rebound					
Electricity	27.0%	11.6%	−10.4%	26.4%	21.2%	47.4%
Non-electricity	30.8%	13.2%	−3.6%	30.6%	24.0%	55.4%

[12] For the algebra of the CES function, and the demonstration that the Cobb–Douglas is a special case of it, see Chiang (1984).

Box 8.5 ***continued***

variables improving – CPI stands for Consumer Price Index. The last two rows show both the commodities electricity and non-electrical energy rebounding – the long-run percentage change in their use is smaller than the initial change in energy efficiency. Note that 27.0% rebound for electricity corresponds to electricity use falling by 3.65%, and 30.8% rebound for non-electricity to its use falling 3.46%.

The calibrated value for the elasticity of substitution between energy and non-energy inputs in the production of the UK composite is 0.3. The results in the third column of Table 8.12 are for a simulation where this elasticity is reduced to 0.1. The macro results are not much affected, but rebound is much reduced. The results in the fourth column are for a simulation where labour costs in each industry are adjusted so as to keep the total cost of production constant (the elasticity of substitution goes back to 0.3, as it does in all of the other simulations). Now the GDP and CPI effects are disadvantageous, and rebound is negative – the eventual energy use reduction is larger than the initial efficiency improvement.

In the Central case government tax receipts increase because of higher earnings and expenditure falls because of reduced social security expenditure, and the government saves the improvement in its fiscal position. For the 'Government expenditure adjusts' simulation, the government spends all of the improvement according to the pattern across sectors from the calibration year. In terms of the variables shown in Table 8.12, this does not make much difference. In the Central case the real wage rate is determined by an equation which has the lagged wage rate and current unemployment as explanatory variables, so that it depends on workers' bargaining power. The last two columns of Table 8.12 treat the labour market differently and at the opposite ends of a spectrum. With 'Exogenous labour supply' the real wage rate moves to clear the labour market, given a fixed supply of labour. At the other extreme, the 'Real wage resistance' simulation has the real wage fixed and the labour market clears by employment variation. As might be expected, the two simulations give quite different macroeconomic variable results. They also give quite different rebound results. Whereas the first gives rebound not very different from in the Central case, the second has it nearly twice as big for non-electricity and more than twice as big for electricity.

Allan *et al.* (2007) do what space does not permit here – tell the stories that explain these results. Like the results, the explanations reflect the model structure. Not everybody would accept that the energy industry structure shown in Figure 8.2, with CES production functions, is a satisfactory model of the UK's energy supply system. As Allen *et al.* note, CGE models are rarely tested for predictive power. In regard to the rebound question, this CGE approach can be compared with that based on input–output which we looked at in Box 8.3. Undoubtedly this approach captures endogenously more of the feedback effects operative in an economy, but the representation of that capture can be problematic.

From the policy perspective, the main message of this study appears to be much the same as that of the study from Box 8.3, and is to be found in the Constant costs column of Table 8.12. Increased efficiency does not have to entail rebound. It can be avoided by accompanying increased efficiency by increased costs, which as in the case of Box 8.3 could take the form of taxation. In fact, policy analysis is rarely concerned with 'manna from heaven' type technical change. Rather, the question is along the lines of 'if an energy tax of £*x* per PJ were introduced, by how much would it improve energy efficiency, and what would its other effects be?'. Even if analysis of 'manna from heaven' technical efficiency improvements indicated 100% rebound, that would not be evidence against policy directed at driving energy efficiency improvements.

8.4.2 The international distribution of abatement costs

Here we consider the CGE modelling work of Whalley and Wigle (1991). In this model the world is divided into six regional economies, as shown in Table 8.13. Two types of energy source are distinguished, the fossil-fuel and other, non-carbon, sources such as nuclear power. These are substitutable for one another in production, and energy and other inputs

Table 8.13 Costs associated with alternative instruments for global emissions reductions

Region	Option 1	Option 2	Option 3
EC	−4.0	−1.0	−3.8
N. America	−4.3	−3.6	−9.8
Japan	−3.7	+0.5	−0.9
Other OECD	−2.3	−2.1	−4.4
Oil exporters	+4.5	−18.7	−13.0
Rest of world	−7.1	−6.8	1.8
World	−4.4	−4.4	−4.2

Source: Adapted from Whalley and Wigle (1991)

are also substitutes in production. International trade involves fossil-fuel energy but not non-fossil energy, and commodities produced using both energy sources. The entries in Table 8.13 are percentage changes in GDP. There is a cost where there is a minus sign, and a gain where there is a plus sign. Given that all fossil fuels are aggregated to a single composite 'fossil fuel', carbon taxation is actually achieved by taxing the fossil-fuel commodity, it being assumed that the only way to reduce emissions is to reduce fuel use.

The results shown in Table 8.13 refer to three alternative routes to the achievement in the model of a global 50% reduction in emissions on what would otherwise have been the case. In options 1 and 2 each economy acts to cut its emissions by 50%. In 1 this is done by the imposition of the required rate of tax on the production of fossil fuels. In 2 it is the consumption of fossil fuels that is taxed. It should be noted that in terms of the discussion of alternative instruments for pollution abatement in Chapter 7, both of these are, at the global level, quantity control, or command and control, type instruments. Each economy is required to cut by 50%, so we are dealing, from the global perspective, with uniform emissions reductions across all sources. Each 'source' in this case is a regional economy, which uses taxation to achieve the emissions cutback required of it. Hence, at the global level standard theory would suggest that neither of these is an efficient way to achieve an overall 50% cut in emissions. This is shown in Table 8.13, where world costs are higher with both 1 and 2 than with option 3, which is the use of fossil fuel taxation at the same rate across all sources. In the model, the uniform global tax is levied and collected by an international agency. In this case it does not make any difference whether the tax is levied on production or consumption.

The results for the individual economies show how the distribution of costs varies with instrument choice. In options 1 and 2, tax revenues accrue to the individual economies, and are spent there. Option 1 then benefits carbon-energy, i.e. fossil-fuel, exporters at the expense of importers, especially the 'rest of world'. Under option 1 GDP increases in the 'oil exporters' economy. The 'rest of world' economy includes the developing nations and the formerly centrally planned economies. It does slightly less badly where the 50% reductions in each economy are achieved by a fossil fuel consumption tax. In this case, the oil exporters suffer heavily, and Japan actually gains. With the tax levied on consumption, the costs to fossil fuel importers are reduced because the pre-tax world price of fossil fuel tax falls, due to reduced demand, and the tax revenues are recycled within the importing economies.

Under option 3 a uniform global tax is levied and collected by an international agency which disposes of the revenues by grants to each economy based on their population size. The per capita grant is the same throughout the world. In this case, not only do we have minimised cost to the global economy, but we also have a distributional impact that works to reduce inequity, by, generally, transferring funds to the 'rest of world' economy, which comprises mainly developing economies. This economy actually gains under option 3 when the joint effect of the tax and revenue distribution is considered. As is clear from the discussion in Chapter 9, equity is important here not only for itself, but also for the incentives for participation that arise. The large developing economies, such as India and China, would gain substantially under option 3. Note that a tradable permits regime could have effects similar to those shown for option 3, if the initial allocation of the permits was arranged so as to favour developing countries. This could be done by doing the initial allocation on the basis of equal per capita shares in the global total of emissions, which was the target, or equivalently in total global fossil fuel use. Each country would get an initial allocation equal to one per capita share times its population size.

8.4.3 Alternative uses of national carbon tax revenue

We now consider some CGE modelling results to illustrate the effects of different assumptions about the way in which any environmental tax revenue is used. Table 8.14 reports results for Australia obtained using a model called ORANI which includes a government sector and Australia's overseas trade. The columns refer to two different simulations of the ORANI model where Australia unilaterally introduces carbon taxation:

S1: Carbon taxation is levied so as to raise revenue of A$2 billion, which is used to reduce payroll taxation by the same amount. The tax rate involved is A$7.40 per tonne of carbon dioxide.

S2: Carbon taxation is levied so as to raise revenue of A$2 billion, which is used to reduce the government deficit by that amount.

The results shown for these two simulations in Table 8.14 are in terms of the percentage differences from the base case without carbon taxation.

In each of these simulations the model standard CGE modelling practice is not followed in that the model is configured with money wage rates fixed, and the market clearing assumption in the labour market is dropped. This allows the model to examine the employment effects of alternative policies. It is regarded as a way of modelling the short-run effects of policy changes. ORANI modellers wishing to examine long-run effects, where 'long run' means enough time for complete adjustment, would set the model with flexible wage rates which would ensure market clearing in the labour market. This illustrates the point that CGE modelling results have to be understood in the light of the assumptions and intentions of the modellers. The same model can produce different results according to the way it is configured for a particular simulation.

The introduction of carbon taxation has an output and a substitution effect in the labour market. There is a reduction in the demand for labour on account of the contraction of economic activity due to fixed money wages and the trade effects of acting unilaterally. There is an increase in the demand for labour on account of the higher price of fossil fuel inputs relative to labour inputs. Where the carbon tax revenue is used to reduce payroll taxation, the non-wage costs to employers of using labour are reduced, thus reinforcing the substitution effect of the carbon tax itself.

This is seen in the comparison of the results for S1 and S2. Employment actually increases where payroll tax is cut in S1. Given the relative shares of labour and fossil fuel in expenditures on inputs, the switch from a tax on labour input to a tax on fossil fuel input leads to a reduction in the consumer price index. The overall impact gives an increase in real GDP. Where, in S2, the carbon tax revenue is used to reduce the budget deficit, employment falls, as does GDP, and the consumer price index increases.

Table 8.14 Effects of carbon taxation according to use of revenue

	S1	S2
Real Gross Domestic Product	0.07	−0.09
Consumer Price Index	−0.18	0.42
Budget Balance*	−0.02	0.31
Employment	0.21	−0.04
CO_2 Emissions	−3.9	−4.7

* As percentage of GDP
Source: Adapted from Common and Hamilton (1996)

8.4.4 Benefits and costs of CGE modelling

As compared with I/O models, CGE models have the benefit that they incorporate behavioural responses to the price changes induced by policy actions on the part of producers and consumers. This entails costs. The structure of the behavioural sub-models reflects the assumptions of economic theory. To economists this is the natural and obvious way to proceed. However, many non-economists would argue that producers and consumers do not actually behave according to those assumptions, and there is quite a lot of evidence that can be cited in support of this view. Economists tend to respond to this line of argument, and the evidence, to the effect that the evidence is flawed and/or that the critics are missing the point, which is that the standard assumptions, and CGE models, are not really about predicting short-run and ephemeral movements but about underlying long-run tendencies. Many non-economists, and some economists, are prone to overlook such caveats and regard CGE models as forecasting models. This appears to be less the case with I/O models, perhaps because their limitations are more readily apparent.

Going from I/O to CGE not only involves more assumptions, but also more data so that those assumptions, or their implications, can be quantified for incorporation into the model. As we have seen, the required data are frequently unavailable, so that the behavioural sub-models often use assumed parameter values that are plausible and consistent with a single benchmark data set for the variables included in the model. CGE model results are sensitive to changes in the parameter values used. Again, this is less of a problem to the extent that such models are seen as vehicles for gaining broad quantitative, or even purely qualitative, insights into policy questions, rather than forecasting models.

The use of CGE, and other economic models, in policy debate is often less fruitful than it might be due to a lack of awareness of the inherent limitations of the models themselves combining with a lack of awareness of the limits to the accuracy with which the variables that they track can actually be measured. The measurement of, for example, national income is not an exact science. The number first produced by a country's official statisticians for GDP is always subject to subsequent revisions. The UK's ONS has published several papers about these revisions, and the 'quality' of its 'estimates' of GDP.[13] In regard to the annual GDP estimates, Mahajan (2006) reports that looking at UK GDP at current prices for 1991 through to 2004, the change between the first estimate published and the latest available at the time of writing ranged from 0.4% to 2.8% of GDP. CGE model results for the national income cost of environmental policies should be looked at with this in mind. In Table 8.13, for example, we noted that the results for the world GDP costs of alternative instruments for a 50% reduction in global emissions did show, as theory predicts, that uniform taxation is the least-cost instrument. Note, however, that the difference between the least cost and the cost with the other instruments is just 0.2%. One might say, then, that while the model confirms the theory, it also suggests that the gain to going for the least-cost approach, rather than the alternatives considered, is quite small. It should be noted also that the model result is not anyway independent confirmation of the least-cost property of uniform taxation – the model incorporates the same assumptions as the standard theory, and could not produce a different result.

Similar considerations apply to the results in Table 8.14, and to those reported in Chapter 9. As regards the exercise reported on in Table 8.14, the real point is the demonstration that a standard economic model says that, in the short run at least, unilateral carbon taxation need not imply an increase in unemployment and a reduction in national income if the revenue is used to reduce a distortionary tax. This is not a point that I/O modelling could make.

Summary

Multi-sectoral models are useful

Economy-wide models take a comprehensive view of the economy as a whole and the interactions between various sectors in the economy. Such models can be used to simulate the consequences, direct, indirect, and induced of shocks or policy changes on the overall economy and its individual sectors.

Extended input–output models can inform environmental policy

Input–output models often have a high level of sectoral disaggregation, permitting the analyst to model changes in a detailed way. When augmented with an environmental sector, or environmental activities,

[13] See, for examples, Richardson (2003), Cook (2004) and Mahajan (2006).

I/O models may be used to investigate economy–environment relationships. In this way, they can form the basis for an investigation of environmental policy options, or for the analysis of the impacts of a variety of 'exogenous' changes, such as changes in technology.

There are no substitution possibilities in input–output models

However, the maintained assumptions that underpin input–output modelling, both in its standard and in its environmentally augmented forms, are very strong, and impose the highly restrictive assumption that there are no substitution possibilities in consumption and production. Any analysis of induced changes in consumption patterns or input uses requires that such are specified exogenously.

CGE models allow for substitution

Computable general equilibrium (CGE) models generalise the I/O framework, in particular by allowing for substitutions to take place in response to changes in relative prices. In recent years CGE models have been employed with increasing frequency as a way of simulating different ways of achieving environmental policy objectives. However, while these models are capable of having a very rich specification, and a theoretically consistent foundation, their usefulness in practice is often limited by difficulty in obtaining appropriate data.

Further reading

Our treatment of input–output analysis and its application to questions concerning natural resources and environmental pollution in the text is simplified though essentially valid. For a comprehensive guide to input–output analysis, including environmental and energy input–output models, see Miller and Blair (1985). Vaze (1998b) presents environmental input–output accounts for the United Kingdom for 1993, and reports the results of simulations based upon them. Proops *et al*. (1993) is an extended account of the use of input–output methods to analyse CO_2 emissions and options for their abatement. Kerkhof *et al.* (2009) is an exercise similar to that of Druckman and Jackson (2009) reported in Box 8.2, but it looks at a wider range of environmental impacts and relates to households in the Netherlands.

Xie (1996) is a good introduction to the use of CGE modelling to study environmental policy issues, and reports results for a Chinese application. Most CGE models are, like the illustrative model discussed in the text, comparative static models. This is a limitation, particularly in the context of a problem like that of thinking about policies for abating CO_2 emissions. The OECD has developed for this problem a dynamic CGE model, which is described and used in Nicoletti and Oliveira-Martins (1993) and Burniaux *et al*. (1994). Hanley *et al.* (2009) use a model closely related to that used in the work reported in Box 8.5, but theirs is for Scotland rather than the UK: a comparison of the Scottish with the UK results is instructive.

GCE models assume optimising behaviour by producers and consumers. There is a lot of evidence that such assumptions are not descriptively accurate. It is, for example, fairly widely accepted that firms are not always technically efficient and that for many of them there exist energy-saving options which would reduce total costs, but which are not adopted. Some economists therefore go in for a kind of hybrid modelling, which uses input–output with econometrically estimated behavioural relationships and more descriptive representations of the energy sector. One such model is the MDM-E3 model developed by Cambridge Econometrics. A useful introduction is Barker *et al.* (2007), which uses the model to look at the effects of the Climate Change Agreements in the UK, according to which firms could pay 80% less under the Climate Change Levy in return for the realisation of agreed CO_2 emissions or energy use reductions. The exercise attributed energy reductions of 2.6% of final energy demand to Climate Change Agreements, including a rebound effect of 19%.

Discussion questions

1. Examine critically the basic assumptions of input–output models, in particular those related to the input–output (Leontief) production function and to factor supplies, and discuss the importance of these assumptions in affecting the validity and accuracy of environmental input–output applications.

2. In discussing the results in Table 8.13 it was asserted that with uniform global taxation, the result would be the same using either fossil fuel production or consumption as tax base. Why is this?

Problems

1. (a) Calculate import, wage and other value-added coefficients from Table 8.1 analogous to the intermediate input coefficients a_{ij}. Check that for each industry, the sum of the intermediate and primary input coefficients is unity.
   (b) In the text we used coefficients derived from the data of Table 8.1 to find the gross output levels implied by new higher levels of export demand. Use those new gross output levels to derive a new transactions table, assuming constant input coefficients for both intermediate and primary inputs. Calculate the new level of GDP.

2. Suppose now we can add the following information about tonnes of waste emissions to the dataset of Table 8.1:

   Industry of origin:

Agriculture	Manufacturing	Services	Total
4000	2500	150	6650

   Calculate the change in the industry and total emissions levels that would be associated with an across-the-board increase of 20% in household expenditure.

3. The transactions table for a closed economy is:

	Agriculture	Manufacturing	Final demand
Agriculture	10	20	50
Manufacturing	20	50	80
Primary inputs	50	80	

   The agriculture industry purchased 10 units of energy, the manufacturing sector 40 units.
   (a) Calculate the energy intensities for deliveries to final demand by the agriculture and manufacturing industries.
   (b) If the use of 1 unit of energy releases 3.5 units of CO_2, calculate total CO_2 emissions and allocate them to deliveries to final demand by the agriculture and manufacturing industries.
   (c) Calculate what total CO_2 emissions would be for deliveries to final demand of 100 from agriculture and 240 from manufacturing.

4. Using the data of Table 8.1, and noting that $(\mathbf{I} - \mathbf{A}')^{-1} = [(\mathbf{I} - \mathbf{A})^{-1}]'$, calculate the effect on prices of a 50% increase in the import costs in the agriculture sector, due to the imposition of a tax on fuel imports. Why is your calculation likely to overestimate the effects of this cost increase?

5. (a) Calculate the real income change that goes with the results given in Box 8.1 for a 50% reduction in emissions.
   (b) If the utility function is assumed to have the form $C_1^{0.5}C_2^{0.5}E^{\delta}$, find the value of δ for which it would be true that utility did not fall with the 50% reduction in emissions.

CHAPTER 9 International environmental problems

The nation-state is here to stay for the near horizon. Thus, practical solutions for today's global challenges must adjust for this reality.

Sandler (1997), p. 212

Learning objectives

During the course of this chapter we address a set of important questions that relate to international environmental problems. After studying this chapter, the reader should understand the implications of these questions, be able to answer them in general terms, and have the ability to apply those general answers to specific international environmental problems.

The questions we deal with are as follows:

- In which ways do international environmental problems differ from purely national (or sub-national) problems?
- What additional issues are brought into contention by virtue of an environmental problem being international?
- What insights does the body of knowledge known as game theory bring to our understanding of international environmental policy?
- What determines the degree to which cooperation takes place between countries and policy is coordinated? Put another way, which conditions favour (or discourage) the likelihood and extent of cooperation between countries?
- Why is cooperation typically a gradual, dynamic process, with agreements often being embodied in treaties or conventions that are general frameworks of agreed principles, but in which subsequent negotiation processes determine the extent to which cooperation is taken?
- Is it possible to use such conditions to explain how far efficient cooperation has gone concerning upper-atmosphere ozone depletion and global climate change?

Introduction

Previous chapters have shown that markets are likely to generate inefficient outcomes in the presence of externalities and public goods. The interdependencies that they create are not, and cannot be, adequately addressed through unregulated market mechanisms. However, when all generators and victims of an externality – or all individuals affected by a public good – reside within a single country, mechanisms exist by which government may be able to induce or enforce an efficient resource allocation where markets fail to do so.[1] These mechanisms can operate because the primacy given to the nation state in political affairs provides the legitimacy and authority needed to support them.

However, many important environmental problems concern public goods or external effects where the affected individuals live (or are yet to live) in many different nation states. These *transboundary environmental problems* are the subject of this chapter. Important examples include global climate change, stratospheric ozone depletion, acid rain, biodiversity loss, and the control of infectious diseases. One

[1] There are two important caveats here. First, where there is widespread dispute about the appropriate boundaries of a nation state, government may not possess the legitimacy required to secure compliance with its regulations. Second, even where it is legitimate, government may have insufficient information or means to achieve efficient outcomes.

property common to these problems is that the level of an environmental cost borne, or benefit received, by citizens of one country does not depend only on that country's actions but also depends on the actions of other countries. Reflect for a moment on these examples. It is evident that in each case the relevant costs and benefits depend on the behaviour of many nations. This adds an additional dimension to environmental analysis. Where environmental impacts spill over national boundaries, or when effective control requires concerted action by groups of countries, there is typically no international organisation with the power to induce or enforce a collectively efficient outcome.[2] Will countries behave selfishly in these circumstances? If so, what will be the consequences of that behaviour? Does mutually beneficial cooperation take place between independent nation states? If it does, how large are the gains from that cooperation? And what can be done to increase the chances of cooperative behaviour? These are the kinds of questions we try to answer in this chapter.

Section 9.1 of this chapter introduces one of the main theoretical tools used to analyse international environmental problems: game theory. Game theoretic analysis has been one of the major developments in the recent environmental economics literature. We begin by looking at a simple two-country, two-strategy game played just once. This model is then generalised to take account of many countries, continuous rather than discrete choices (for example, *how much* pollution abatement rather than *whether or not* to abate pollution), and games played repeatedly rather than just a single time.

Game theory is applied here in the context of decisions about the provision of an international public good. This focus is taken because many – if not most – transboundary environmental problems concern the provision (or maintenance) of public goods, and because the theory of public goods has underpinned much of the recent literature. Thus much of the content of this chapter deals with public goods externalities that spill over national boundaries.

Sections 9.2 and 9.3 examine international environmental agreements (IEA), the main institutional arrangement that has been used in attempts to obtain cooperative solutions to international environmental problems. The techniques of game theory provide a foundation for our analysis of IEAs. They throw light on what factors influence the degree of effectiveness of an IEA, and can contribute towards the development of a set of design principles for 'good' IEAs.

The next two sections investigate two important international environmental problems: depletion of the stratospheric ozone layer (in 9.4) and global climate change (in 9.5). These outline the relevant science, consider the likely costs and benefits of policy intervention, and discuss implications for international environmental policy.

We end the chapter by drawing together some of the conclusions of our analyses. The chapter is not exhaustive in its treatment of international environmental problems. Several other such problems that are primarily related to land-use change are taken up in other parts of this book. In particular, tropical deforestation, wilderness conversion and the loss of biological diversity are left for consideration in Chapters 17 and 18 in our examination of the economics of renewable resources. Other issues, including that of acid rain, are examined in separate documents on the Companion Website.

9.1 Game theory analysis[3]

A powerful technique for analysing behaviour where actions of individuals or firms are interdependent is

[2] One might argue that international law provides an institutional framework that provides such an enforcing mechanism. But this position is not really tenable. The International Court of Justice (part of the United Nations system) is the principal institution for using international law to rule on disputes between nations. The official website of the ICJ states that: 'The Court can only deal with a dispute when the States concerned have recognized its jurisdiction. No State can therefore be a party to proceedings before the Court unless it has in some manner or other consented thereto.' See http://www.icj-cij.org/homepage/

[3] This section draws heavily on the works of two writers: Todd Sandler and Scott Barrett. Sandler's book *Global Challenges* (1997) is a superb non-technical account of game theory applied to international environmental problems. The sequencing and much of the content of our game theory arguments owe much to the pieces by Barrett listed in the Further Reading to this chapter. The chapter has also drawn extensively from an excellent review of international environmental agreements by Michael Finus (2002).

The games we shall consider here are played by two or more countries. Another variant of game theory, not considered in this chapter, is known as 'games against nature', which is concerned with choices by just one player under conditions of uncertainty. However, in Chapter 13 we use these to consider decision making in the face of uncertainty.

game theory. We shall make extensive use of this technique to investigate behaviour in the presence of global or regional public goods. The arguments also apply to externalities that spill over national boundaries.

Game theory as it is typically used assumes that players are concerned only with the returns that they get. This is, of course, a very strong assumption, and one that readers may not be willing to accept. In Section 9.3.5 we shall briefly return to this matter to consider some of the reasons why one might consider it to be an inappropriate assumption. For now, though, we will work with this assumption imposed.

Game theory is used to analyse choices where the outcome of a decision by one player depends on the decisions of the other players, and where decisions of others are not known in advance. This interdependence is evident in environmental problems. Where pollution spills over national boundaries, expenditures by any one country on pollution abatement will give benefits not only to the abating country but to others as well. Similarly, if a country chooses to spend nothing on pollution control, it can obtain benefits if others do so. So in general the pay-off to doing pollution control (or not doing it) depends not only on one's own choice, but also on the choices of others.

9.1.1 Two-player binary-choice games

We investigate first some simple two-player games. In the variants considered here, each player (*X* and *Y*) faces a binary choice: select either Strategy 1 or Strategy 2. Strategy choices are made simultaneously, so that neither player can observe the strategy selected by the other player before making his or her own choice. The game is played just once. The elements of these games can be represented in the generic form shown in Figure 9.1.

The pair of letters in each cell of the matrix denotes the net benefit (or 'pay-off') that each country receives for a particular choice of strategy by *X* and by *Y*. The first letter denotes the pay-off to *X*, the second the pay-off to *Y*. When the letters in Figure 9.1 are replaced by numerical values, a particular form of game is generated, and we can explore its outcome. As there are many alternative structures of pay-off matrix, there is a large number of possible game forms. We begin with a game that has been widely used to analyse international environmental problems: the *Prisoner's Dilemma*.

9.1.1.1 A Prisoner's Dilemma game

This was first introduced in Chapter 4. Suppose now that *X* and *Y* are two countries, each of which faces a binary choice of whether to abate pollution (labelled '*Abate*') or not to abate pollution (labelled '*Pollute*'). Pollution abatement is assumed to be a public good so that abatement by either country benefits both. Abatement comes at a cost of 7 to the abater, but confers benefits of 5 to both countries. If both abate both experience benefits of 10 (and each experiences a cost of 7).

The pay-offs to each country from the four possible configurations of strategy choices are shown in Figure 9.2. We have assumed that prior to the game being played neither country was abating its pollution. Hence, if both countries select *Pollute*, neither incurs any additional cost or benefit relative to the pre-game baseline, and so the net benefit to each is 0. Other pay-offs can be deduced from the information given above, and in each case are expressed relative to the pre-game baseline. For example, if *X* chooses *Abate* and *Y Pollute* the pay-offs are –2 to *X* (a benefit of 5 minus a cost of 7) and 5 to *Y* (a benefit of 5 at no cost).

Y's strategy / *X*'s strategy	*Strategy 1*	*Strategy 2*
Strategy 1	a, a	b, c
Strategy 2	c, b	d, d

Figure 9.1 Two-player binary choice games

Y's strategy / *X*'s strategy	*Pollute*	*Abate*
Pollute	**0, 0**	5, –2
Abate	–2, 5	3, 3

Figure 9.2 A two-player pollution abatement game

9.1.1.1.1 The non-cooperative solution

To predict the outcome of this game, it is necessary to consider how the countries handle their strategic interdependence. We investigate the consequences of two different ways in which interdependence may be handled. The first approach is to assume that each country maximises its own net benefit, conditional on some expectation about how the other will act, and without collaboration taking place between the countries. We describe this kind of behaviour as 'non-cooperative'. If this leads to an equilibrium outcome, that outcome is called a non-cooperative solution to the game. Alternatively, 'cooperative behaviour' takes place when countries collaborate and make agreements about their strategic choices. If an equilibrium outcome exists, it is called a cooperative solution to that game.

We begin by looking at non-cooperative behaviour. One helpful concept used in looking for solutions to non-cooperative games is the idea of *dominant strategy*. A player has a dominant strategy when it has one strategy that offers a higher pay-off than any other irrespective of the choice made by the other player. A widely accepted tenet of non-cooperative game theory is that dominant strategies are selected where they exist.

Let us examine the pay-off matrix to see whether either country has a dominant strategy. First, look at the game from *Y*'s point of view. *Y* can reason as follows:

- I do not know whether *X* will *Pollute* or *Abate*. But if *X* were to choose *Pollute*, my preferred choice would be *Pollute*, as the pay-off to me of not abating, 0, exceeds my pay-off from abating, –2.
- I do not know whether *X* will *Pollute* or *Abate*. But if *X* were to choose *Abate*, my preferred choice would be *Pollute*, as the pay-off to me of not abating, 5, exceeds my pay-off from abating, 3.
- For whichever choice *X* does actually make, my best 'response' would be to select *Pollute*.

We see that whatever *X* chooses, *Pollute* is best for *Y*, and so is *Y*'s dominant strategy. You should confirm that the dominant strategy for *X* is also *Pollute*.[4] Both countries have a dominant strategy; and for both of them that strategy is *Pollute*. Given that dominant strategies will be played where they exist, non-cooperative game theory analysis leads to the conclusion that the equilibrium solution to this game consists of both countries not abating pollution.

It is worth remarking on three characteristics of this solution. First, the fact that neither country chooses to abate pollution implies that the state of the environment will be worse than it could be. Second, the solution is also a *Nash equilibrium*.[5] A set of strategic choices is a Nash equilibrium if each player is doing the best possible given what the other is doing. Put another way, neither country would benefit by deviating unilaterally from the outcome, and so would not unilaterally alter its strategy given the opportunity to do so. Third, the outcome is inefficient. Both countries could do better if they had chosen to abate (in which case the pay-off to each would be three rather than zero).

Why has this state of affairs come about? There are two facets to the answer. The first is that the game has been played non-cooperatively. We shall examine shortly how things might be different with cooperative behaviour. The second concerns the pay-offs used in Figure 9.2. These pay-offs determine the structure of incentives facing the countries. They reflect the assumptions we made earlier about the costs and benefits of pollution abatement. In this case, the incentives are not conducive to the choice of abatement.

Not surprisingly, the structure of incentives can be crucial to the outcome of a game. The pay-off matrix in Figure 9.2 is an example of a so-called

[4] One could confirm this by repeating the reasoning used above, but this time from *X*'s point of view. However, as the structure of this pay-off matrix is symmetric, so that the labelling *X* and *Y* could be reversed with no loss of information, it is not necessary in this case to repeat the calculation.

[5] This term is named for John Nash who was awarded a Nobel Prize in 1994 for his contributions to game theory, and was the subject of the film *A Beautiful Mind* which won four Academy Awards.

Y's strategy / X's strategy	*Pollute*	*Abate*
Pollute	**2, 2**	4, 1
Abate	1, 4	3, 3

Figure 9.3 The two-player pollution abatement Prisoner's Dilemma game: ordinal form

Prisoner's Dilemma game. The Prisoner's Dilemma is the name given to all games in which the pay-offs, when put in ordinal form, are as shown in Figure 9.3. The ordinal form of the pay-off matrix ranks pay-offs rather than showing them in cardinal values. In Figure 9.3, the number 1 denotes the least preferred pay-off and 4 the most preferred pay-off. Take a moment to confirm to yourself that the rankings in Figure 9.3 do correspond to the cardinal pay-offs in Figure 9.2.

In *all* Prisoner's Dilemma games, there is a single Nash equilibrium (the outcome highlighted in bold in Figures 9.2 and 9.3). This Nash equilibrium is also the dominant strategy for each player. Moreover, the pay-offs to both countries in the dominant strategy Nash equilibrium are less good than those which would result from choosing their alternative, dominated strategy. As we shall see in a moment, not all games have this structure of pay-offs. However, so many environmental problems appear to be examples of Prisoner's Dilemma games that environmental problems are routinely described as Prisoner's Dilemmas.

9.1.1.1.2 Cooperative solution

Suppose that countries were to cooperate, perhaps by negotiating an agreement. Would this alter the outcome of the game? Intuition would probably lead us to answer yes. If both countries agreed to abate – and did what they agreed to do – pay-offs to each would be 3 rather than 0. So in a Prisoner's Dilemma cooperation offers the prospect of greater rewards for both countries, and in this instance superior environmental quality.

But this tentative conclusion is not robust. Can these greater rewards be sustained? If self-interest governs behaviour, they probably cannot. To see why, note that the {*Abate, Abate*} outcome is not a Nash equilibrium. Alternatively, and to anticipate a concept that will be explained later, this agreement is not 'self-enforcing'. Each country has an incentive to defect from the agreement – to unilaterally alter its strategy once the agreement has been reached. Imagine that the two countries are at the cooperative solution. Now look at the incentives facing *Y*. Given that *X* has chosen to abate, *Y* can obtain an advantage by defecting from the agreement ('free-riding'), leaving *X* to abate but not abating itself. In this way, country *Y* could obtain a net benefit of 5 units. Exactly the same argument applies to *X*, of course. There is a strong incentive operating on each player to attempt to obtain the benefits of free-riding on the other's pollution abatement. These incentives to defect from the agreement mean that the cooperative solution is, at best, an unstable solution.

9.1.1.1.3 A binding agreement?

Is it possible to transform this game in some way so that the {*Abate, Abate*} strategy pair becomes a stable cooperative solution? There are ways in which this might be done, several of which we shall examine later in the chapter. One possibility would be to negotiate an agreement with built-in penalty clauses for defection. For example, the agreement might specify that if either party defects (pollutes) it must pay a fine of 4 to the other. If you construct the pay-off matrix that would correspond to this agreement, it will be seen that the game structure has been transformed so that it is no longer a Prisoner's Dilemma game. Moreover, both countries would choose to abate.

But we should be hesitant about accepting this conclusion. Countries might make such promises but, given the incentive to defect, would they keep them or could they be made to keep them? As we have seen there is an incentive to renege on promises here. Cheating (or reneging or free-riding) on agreements might confer large gains on individual cheaters, particularly if the cheating is not detectable. And countries could only be forced to keep their promises (or pay their fines) if there were a third party who could enforce the agreement. So to secure the collectively best outcome, and to make the agreement binding in a game-theory rather than legal sense, it would seem that an enforcer is required. But in a world of sovereign states, no such enforcer exists. So it would seem to be the case that agreements between nations must be *self-enforcing* if they are to be sustained.[6] The only self-enforcing equilibrium here seems to be the non-cooperation outcome.

All of this suggests that cooperation cannot be relied upon to prevail in a situation characterised by individual countries acting non-cooperatively in ways which they perceive to be in their own interests. Non-cooperative outcomes can and do happen, even where it would be in the interest of all to behave cooperatively. It is for this reason that the game we have been discussing was called a dilemma. Players acting in an individually rational way end up in a bad state. If they attempt to collaborate, incentives on the other to cheat on the deal expose each to the risk of finishing up in the worst of all possible states. Mutual defection seems to be inevitable where the structure of pay-offs has the form of a Prisoner's Dilemma and the game is a one-shot game (it is played only once).

Fortunately, not all games have the structure of the Prisoner's Dilemma (PD). And even where a game does have a PD pay-off matrix structure, the game may be played repeatedly. As we shall see later, repetition substantially increases the likelihood of cooperative outcomes being obtained. Furthermore, there *may* be ways in which a PD game could be successfully transformed to a type that is conducive to cooperation.

9.1.2 Other forms of game

Not all games have the form of the Prisoner's Dilemma. Indeed, Sandler (1997) states that there are 78 possible ordinal forms of the pay-off matrix for a one-shot, 2-player, 2-strategy game, found from all the permutations of the rankings 1 through to 4. Two other structures of pay-off matrix appear to be highly relevant to environmental problems. These structures generate the Chicken game and the Assurance game, each of which is useful in exploring aspects of international environmental problems.

9.1.2.1 Chicken game

Let us revisit the previous PD game in which each of two countries must choose whether or not to abate pollution. We suppose, as before, that each unit of pollution abatement comes at a cost of 7 to the abater and, being a public good, confers benefits of 5 to both countries. However, in the game considered now, it is assumed that doing nothing exposes both countries to serious pollution damage, at a cost of 4 to both countries. The pay-off matrix for this 'Chicken game' is presented in Figure 9.4.[7] The only difference between this set of pay-offs and that in Figure 9.2 is the negative (as opposed to zero) pay-offs in the cell corresponding to both countries selecting *Pollute*.

This difference fundamentally changes the nature of the game.[8] Consider, first, non-cooperative behaviour. Neither player has a dominant strategy. Moreover, there are two Nash equilibria (the cells in bold). Game theory predicts that in the absence of a dominant strategy equilibrium, a Nash equilibrium will be played (if one exists). Here there are two Nash equilibria (bottom left and top right cells), so there is some indeterminacy in this game. How can this indeterminacy be removed? One possibility arises from commitment or reputation. Suppose that *X* commits herself to pollute, and that *Y* regards this commitment as credible. Then the bottom row of the matrix becomes irrelevant, and *Y*, faced with

Y's strategy / *X*'s strategy	*Pollute*	*Abate*
Pollute	–4, –4	**5, –2**
Abate	**–2, 5**	3, 3

Figure 9.4 A two-player Chicken game

[6] Later in this chapter we shall give a precise explanation of what is meant by a self-enforcing agreement.

[7] The description 'Chicken game' comes from the fact that this form of pay-off matrix is often used to describe a game of daring in which two motorists drive at speed towards each other. The one who loses nerve and swerves is called Chicken. Relabelling the strategy *Pollute* as *Maintain Course* and *Abate* as *Swerve* generates a plausible pay-off matrix for that game.

[8] If you transform the Chicken game pay-off matrix into its ordinal form, you will see that the difference in ordinal forms of the Prisoner's Dilemma and the Chicken game lies in the reversal of the positions in the matrices of the 1 and 2 rankings.

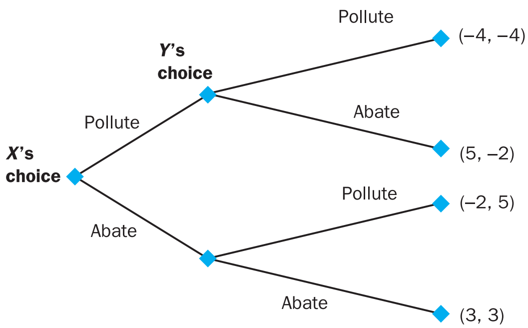


Figure 9.5 Extensive form of the Chicken game

pay-offs of either −2 or −4, will choose to abate. (We may recognise this form of behaviour in relationships between bullies and bullied.) Another possibility arises if the game is played *sequentially* rather than simultaneously. Suppose that some circumstance exists so that country *X* chooses first.[9] *Y* then observes *X*'s choice and decides on its own action. In these circumstances, the *extensive form* of the game is relevant for analysis of choices. This is illustrated in Figure 9.5. The solution to this game can be found by the method of backward induction. If *X* chooses *Pollute*, *Y*'s best response is *Abate*. The pay-off to *X* is then 5. If *X* chooses *Abate*, *Y*'s best response is *Pollute*. The pay-off to *X* is then −2. Given knowledge about *Y*'s best response, *X* will choose *Pollute* as her pay-off is 5 (rather than −2 if she had selected *Abate*). This is one example of a more general result: in games where moves are made sequentially, it is *sometimes* advantageous to play first – there is a 'first-mover advantage'. First-mover advantages exist in Chicken games, for example.[10]

Now consider another possibility. Suppose there is asymmetry in the top left cell so that the penalty to *X* of not abating is −1 rather than −4, but all else remains unchanged. (This is no longer a Chicken game, however, as can be seen by inspecting the ordinal structure of pay-offs.) The outcome of this game is determinate, and the strategy combination corresponding to the top right cell will be chosen. Backward induction shows that *X* has a dominant strategy of *Pollute*. Given that *Y* expects *X* to play her dominant strategy, *Y* plays *Abate*.[11]

This is reminiscent of decisions relating to ozone-layer depletion. For a while, at least, some countries expected the USA to reduce ozone-depleting emissions, and were content to free-ride on this. Indeed, the USA did play a major role in leading the way towards reducing their usage of ozone-depleting substances. Two reasons seemed to lie behind this. First, US EPA studies published in 1987 had shown that health costs from ozone depletion were dramatically higher than control costs. (Specifically, a 50% cut in CFC emissions was estimated to create long-term benefits in the form of avoided cancer damage valued at $6.4 trillion, while long-run abatement costs would be in the range $20–40 billion.[12]) Second, chemical businesses in the USA were confident of being able to achieve competitive advantage in the production of substitute products to CFC substances. The USA would be in a very strong position were a CFC ban to be introduced. Overall, the USA perceived that the benefits to her of abatement were high relative to the benefits of not abating. This was not true for all countries, however, and it is this that creates an asymmetry in the pay-off matrix. Those countries which were, initially at least, free-riders had less relative advantage in abating.

Cooperative behaviour in the Chicken game?

A strategy in which both countries abate pollution could be described as the 'collectively best solution' to the Chicken game as specified in Figure 9.4; it maximises the sum of the two countries pay-offs. But that solution is not stable, because it is not a Nash equilibrium. Given the position in which both countries abate, each has an incentive to defect (provided the other does not). A self-enforcing agreement in which the structure of incentives leads countries to negotiate an agreement in which they

[9] A commitment or a reputation might be interpreted in this way. That is, the other player (in this case *Y*) regards *X* as already having made their choice of strategy.

[10] However, some other structures of pay-off matrix lead to the opposite result, in which it is better to let the other player move first and then take advantage of that other player.

[11] Dixit and Nalebuff (1991) give a more complete account of the reasoning that lies behind strategic choices in these kinds of games.

[12] Some later (1989) US EPA estimates are presented in Table 9.4. While the estimates are rather different, the huge surplus of benefits over costs remains in the later figures.

will all abate *and* in which all will wish to stay in that position once it is reached does not exist here. However, where the structure of pay-offs has the form of a Chicken game, we expect that *some* protective action will take place. Who will do it, and who will free-ride, depends on particular circumstances. Leadership by one nation (as by the USA in the case of CFC emissions reductions) may be one vehicle through which this may happen.

9.1.2.2 Assurance game

The other game-form to which some attention will be given in this chapter is the Assurance game. We consider this in terms of an example in which each of two countries must decide whether or not to contribute to a global public good. The cost to each country of contributing is 8. Benefits of 12 accrue to each country only if both countries contribute to the public good. If one or neither country contributes there is no benefit to either country. What we have here is a form of threshold effect: only when the total provision of the public good reaches a certain level (here 2 units) does any benefit flow from the public good. Situations in which such thresholds exist may include the control of infectious disease, conservation of biodiversity, and the reintroduction of species variety into common-property resource systems. The pay-off matrix which derives from the cost and benefit values described above is given in Figure 9.6.

Inspection of the pay-off matrix reveals the following. Looking first at non-cooperative behaviour, neither country has a dominant strategy. There are two Nash equilibria (shown in bold in the matrix). Which is the more likely to occur? Perhaps surprisingly, game theory cannot be of much help in answering that question for a single-shot game. However (as we show later), if the game were to be played repeatedly there is a strong presumption that both would contribute. Moreover, the greater is the difference between the pay-offs in the two Nash equilibria, the more likely is it that the 'both countries contribute' outcome will result. The cooperative solution is that in which both contribute. This solution is stable because it is self-enforcing. If one player cooperates, it is in the interest of the other to do so too. Once here, neither would wish to renege or renegotiate. The incentive structure here is supportive of cooperation.

B's strategy / A's strategy	*Do not contribute*	*Contribute*
Do not contribute	**0, 0**	0, –8
Contribute	–8, 0	**4, 4**

Figure 9.6 A two-player Assurance game

9.1.3 Games with multiple players

The analysis so far has been restrictive, as it has involved only two-country games. But most international environmental problems involve several countries and global problems a large number. However, much of what we have found so far generalises readily to problems involving more than two countries. Let N be the number of countries affected by some environmental problem, where $N \geq 2$. For simplicity, we assume that each of the N countries is identical.

We begin by revisiting the Prisoner's Dilemma example, discussed first in Section 9.1.1.1. As before, each unit of pollution abatement comes at a cost of 7 to the abating country; it confers benefits of 5 to the abating country and to all other countries. For the case where $N = 10$, the pay-off matrix can be described in the form of Table 9.1.

Here we look at things from the point of view of one country, the ith country let us say. Table 9.1 lists the pay-off to country i from polluting and from abating for all possible numbers of countries **other than** i that choose to abate. For each column (number of countries other than country i abating) nation i does better by polluting rather than by abating. It is evident that irrespective of how many other countries decide to abate, it is individually rational for country i to not abate. Given that all countries are identical, and so i could refer to any of the N countries, the non-cooperative solution is *Not Abate* by all 10 countries. The basic properties of the two-country Prisoner's Dilemma are again evident. Nations following their self-interest each finish up with worse outcomes (0 each) than if all were to cooperate and abate pollution (43 each). But the

Table 9.1 The Prisoner's Dilemma example with 10 countries

	Number of abating nations other than *i*									
	0	1	2	3	4	5	6	7	8	9
Nation *i* pollutes	0	5	10	15	20	25	30	35	40	45
Nation *i* abates	–2	3	8	13	18	23	28	33	38	43

cooperative solution remains unstable. Given an agreement to *Abate*, any individual country does better by reneging on the agreement and polluting.

As before, the structure of pay-offs is critical in determining whether cooperation can be sustained. To explore this idea further, let us think about the pay-offs to choices in a more general way than we have so far. Following Barrett (1994a), we denote NB_A as the net benefit to a country if it abates and NB_P as the net benefit to a country if it pollutes (does not abate). Let there be N identical countries, of which K choose to abate. We define the following pay-off generating functions:

$$NB_P = a + bK; \; NB_A = c + dK$$

where a, b, c and d are parameters. By altering these parameter values, we generate different pay-off matrices. For the problem in Table 9.1 we have $a = 0$, $b = 5$, $c = -7$ and $d = 5$. You should verify that these two expressions do indeed generate the numbers shown in the table. Note that for the 'Nation *i* pollutes' row in Table 9.1 K is equal to the 'Number of abating nations other than *i*', whereas for the 'Nation *i* abates' row K is equal to the 'Number of abating nations other than *i*' plus one.

It will be convenient to portray the information shown in Table 9.1 in another way – in the form of Figure 9.7. You should now examine Figure 9.7, and verify that this also represents the information correctly. (If you wish to see the calculations that lie behind this chart, look at the Excel file *games.xls*.) It is again clear from this chart that the net benefit of pollution is always larger than the net benefit of abating, irrespective of how many other countries abate. The only stable outcome is that in which no countries abate.

But this conclusion is *not* true for all pay-off structures. For example, suppose that the parameters of the pay-off functions take the following values: $a = 12$, $b = 3$, $c = -7$ and $d = 7$. Then if we generate the counterparts to Table 9.1 and Figure 9.7, we obtain Table 9.2 and Figure 9.8.

Table 9.2 The Prisoner's Dilemma example with alternative parameter values

	Number of abating nations other than *i*									
	0	1	2	3	4	5	6	7	8	9
Nation *i* pollutes	12	15	18	21	24	27	30	33	36	39
Nation *i* abates	0	7	14	21	28	35	42	49	56	63

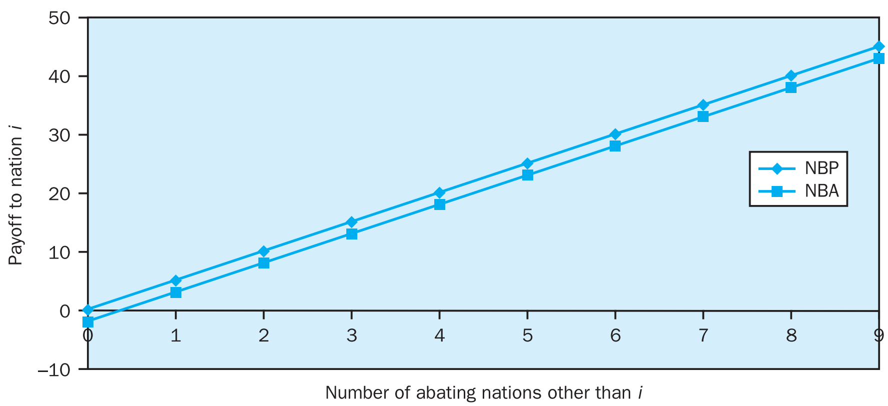


Figure 9.7 The pay-offs to one country from abating and from not abating as the number of other countries abating varies

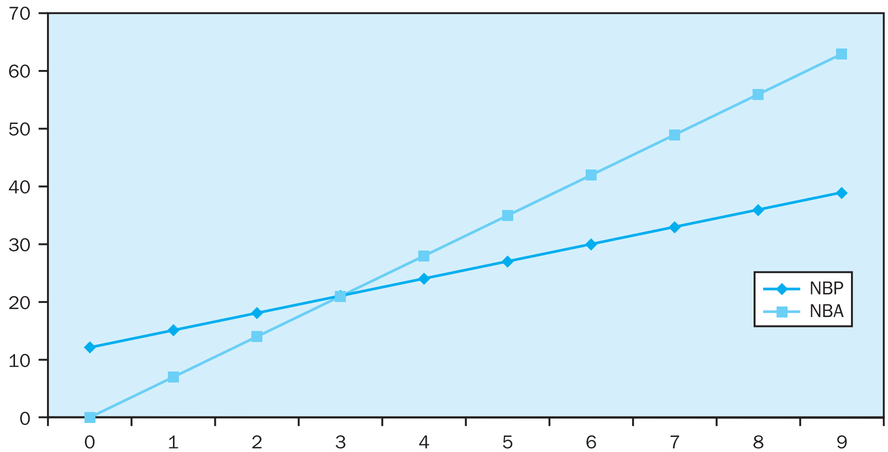


Figure 9.8 The pay-offs to one country from abating and from not abating as the number of other countries abating varies: alternative set of parameter values

It is evident from either of these two descriptions that if less than three countries agree to cooperate (abate), none will cooperate. That is, all will pollute. However, if three or more cooperate, all will cooperate. Here we have an outcome in which two stable equilibria are possible: either all will abate, or none will. To see that they are both stable equilibria, reason as follows. First, suppose that no country abates. Then, can any country individually improve its pay-off by abating rather than polluting? The answer is no. Next, suppose that every country abates. Can any country individually improve its pay-off by polluting rather than abating? The answer is again no. However, no other combination of polluting and abating countries is stable. For example, suppose that you are polluting and two other countries are abating (and so the remaining seven pollute). You get 18 and they get 14 each. But each of the two abaters has an incentive to defect (i.e. pollute). For if one were to do so, its pay-off would rise from 14 to 15. Verify that this is so.

As a third example, now consider the parameter set $a = 0$, $b = 5$, $c = 3$ and $d = 3$. This is represented in Table 9.3 and Figure 9.9. This structure of pay-offs generates an incomplete self-enforcing agreement with 3 signatories and 7 non-signatories. Notice that the pay-off to each cooperating (abating) country is lower than that to each non-cooperating country. In this respect the game is similar to a Chicken game. The collective pay-off to all countries is greater than where no cooperation takes place, but is less than from complete cooperation. In this respect, the game is reminiscent of a Prisoner's Dilemma that has been partly solved. The lesson of this story is that if a Prisoner's Dilemma pay-off matrix can be transformed by altering the structure of pay-offs (so that, for example, it resembles one of the two later examples) stable cooperation becomes possible. But cooperation may still be less than complete. We will return to this theme in a short while. Before we do so, one further generalisation is necessary.

Table 9.3 The Prisoner's Dilemma example with third set of parameter values ($a = 0$, $b = 5$, $c = 3$ and $d = 3$)

	Number of abating nations other than *i*									
	0	1	2	3	4	5	6	7	8	9
Nation *i* pollutes	0	5	10	15	20	25	30	35	40	45
Nation *i* abates	6	9	12	15	18	21	24	27	30	33

9.1.4 Continuous choices about the extent of abatement

Our discussion so far has been rather limiting as we have assumed that nations face a simple binary choice decision: participate in an environmental agreement and abate pollution, or do not participate in the

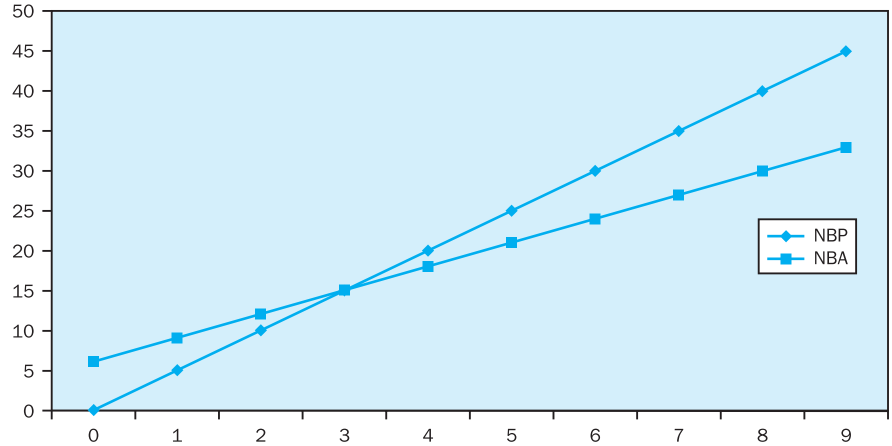


Figure 9.9 The pay-offs to one country from abating and from not abating as the number of other countries abating varies: third set of parameter values ($a = 0$, $b = 5$, $c = 3$ and $d = 3$)

agreement and do not abate. But in practice, the relevant decision is not an all-or-nothing choice. Even if one chooses to participate in an agreement, there is a further choice to make: by how much should that country agree to abate the pollutant. Let us now generalise the discussion by allowing countries to choose – or rather negotiate – abatement levels.

This can be done with some simple algebra. Our previous analysis has shown that in terms of the participation choice, three kinds of outcome are possible: none abate, all abate, and some abate but others do not. For simplicity, we deal first with just the first two of these alternatives. Let us assume that there are N identical countries, indexed by $i = 1, \ldots, N$. We first look at each country's pay-off function.

The pay-off functions

Each country is taken to maximise some net benefit (or pay-off) function, Π_i. Let z_i denote pollution abatement by country i, and $Z = \sum_{i=1}^{N} z_i$ be the total abatement of the pollution. Pollution abatement is a public good. Then the pay-off (or net benefit) of abatement to country i is the benefits of abatement, B_i (which depends on the total amount of abatement by *all* countries) minus costs to country i of abatement, C_i (which depends on its *own* level of abatement). Thus we have

$$\Pi_i = B_i(Z) - C(z_i), \quad \text{for } i = 1, \ldots, N \tag{9.1}$$

It is important to be aware of the use of subscripts in this equation. There is no i subscript on C because we are continuing to assume that all countries are identical and so the cost function itself does not vary between countries. However, the i subscript in B_i signifies that even though the argument of B_i is *all* countries emissions reductions, it is only country i's benefits that are included in equation 9.1.

9.1.4.1 Non-cooperative behaviour

Non-cooperative (or unilateral) behaviour involves each country choosing its level of abatement to maximise its pay-off, independently of – and without regard to the consequences for – other countries. That is, each country chooses z to maximise equation 9.1 conditional on z being fixed in all other countries.

Country i's abatement choice is the solution to the first-order condition

$$\frac{\mathrm{d}B_i(Z)}{\mathrm{d}Z}\frac{\mathrm{d}Z}{\mathrm{d}z_i} = \frac{\mathrm{d}C(z_i)}{\mathrm{d}z_i} \tag{9.2}$$

Noting that $\mathrm{d}Z/\mathrm{d}z_i = 1$, and that, given our assumption of symmetry, all countries' efficient abatement will be identical, the solution can be written as

$$\frac{\mathrm{d}B_i(Z^U)}{\mathrm{d}Z} = \frac{\mathrm{d}C(z^U)}{\mathrm{d}z} \quad \text{where } Z^U = \sum_{i=1}^{N} z^U \tag{9.3}$$

and the superscript U denotes the unilateral (non-cooperative) solution. Intuitively, each country abates up to the point where its own marginal benefit of abatement is equal to its marginal cost of abatement.

9.1.4.2 Full cooperative behaviour

Full cooperative behaviour consists of the N countries *jointly* choosing levels of abatement to maximise their collective pay-off. This is equivalent to what would happen if the N countries were unified as a single country that behaved rationally.[13] The solution requires that abatement in each country be chosen jointly to maximise the collective pay-off

$$\Pi = NB_i(Z) - \sum_{i=1}^{N} C(z_i)$$

The necessary conditions for a maximum are

$$N\frac{\mathrm{d}B_i(Z)}{\mathrm{d}Z}\frac{\mathrm{d}Z}{\mathrm{d}z_i} = \frac{\mathrm{d}C(z_i)}{\mathrm{d}z_i} \quad \text{for all } i.$$

Once again (for the same reasons as given earlier) these can be written as

$$N\frac{\mathrm{d}B_i(Z^C)}{\mathrm{d}Z} = \frac{\mathrm{d}C(z^C)}{\mathrm{d}z} \quad \text{where } Z^C = \sum_{i=1}^{N} z^C$$

where the superscript C denotes the full cooperative solution. This is the usual condition for efficient provision of a public good. That is, in each country, the marginal abatement cost should be equal to the sum of marginal benefits over all recipients of the public good. The full cooperative solution can be described as collectively rational: it is welfare-maximising for all N countries treated as a single entity. Indeed, if some supranational governmental body existed, acting to maximise total net benefits, and had sufficient authority to impose its decision, then the outcome would be the full cooperative solution described here.

The non-cooperative and cooperative solutions can be visualised graphically, and are represented in Figure 9.10. The diagram is adapted from Barrett (1994a). Z denotes pollution abatement. In the absence of cooperation, equilibrium abatement is Z_N. Here, each country equates its own marginal benefit of abatement (MB_i) and marginal cost of abatement (MC_i). In contrast, the full cooperation abatement level Z_C has each country equating the sum of the marginal benefit of abatement across all countries (MB) with its own and marginal cost of abatement. This picture is useful because it shows us what determines the size of two magnitudes of interest:

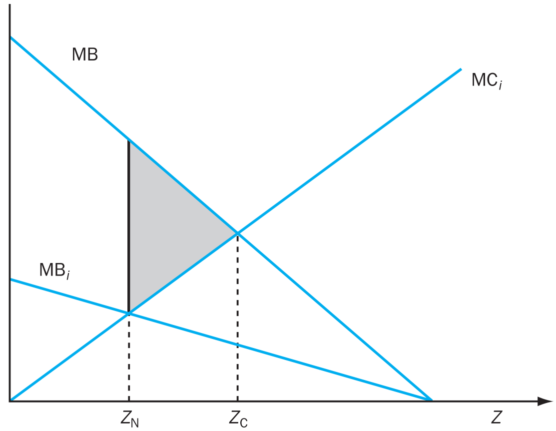


Figure 9.10 A comparison of the non-cooperative and full cooperative solutions to an environmental public good problem

- the amount by which full cooperation abatement exceeds non-cooperative abatement (i.e. $Z_C - Z_N$);
- the magnitude of the efficiency gain from full cooperation (the shaded triangular area in the diagram).

It is evident that these depend on two things:

1. the relative slopes of the MB_i and MC_i curves;
2. the number of competing countries, N (as this determines the relative slopes of the MB_i and MB curves.

Problem 9.2 invites you to examine these matters further, and to draw inferences about the conditions under which international cooperation is likely to deliver large decreases in emissions.

[13] The joint decision process may also involve negotiations about how the additional benefits from cooperation are to be distributed between the parties but we shall leave this matter for consideration later.

9.2 International environmental agreements

9.2.1 International environmental agreements as a vehicle for international cooperation

One indicator of the extent to which international environmental cooperation has taken place is the number of international environmental agreements (IEA) in existence.[14] Moreover, the rate at which new agreements are formed may also serve as an indicator of the intensity of such cooperation over some interval of time. The number and range of IEAs have grown tremendously in recent years. Estimates of the total number of such treaties vary depending on how broadly the term is defined. For example, Keohane *et al.* (1993) estimated there to be 140 international environmental agreements instituted since between 1921 and 1992, with more than half entered into force after 1972. Brown Weiss *et al.* (1992), including regional and bilateral treaties in their count, arrive at the much higher number of 900 agreements in force in 1992. It seems safe to say that there are now several hundred international environmental agreements.

Many of the early environmental treaties concerned regulation of behaviour at sea: marine fishing (see Chapter 17), transportation in international waters, dumping and disposal of wastes, and exploitation of the sea beds. During the second half of the twentieth century, a substantial proportion of new agreements related to bilateral or regional pollution spillover problems. Since the mid-1980s, great attention has been paid to attempts to develop agreements about the use of two global public goods: composition of the atmosphere and the stock of biological diversity.[15]

The main vehicle that has been used in attempts to reach cooperative solutions to regional and global environmental problems is the intergovernmental conference. The proactive role played by the United Nations (UN) system of international institutions has been one of the successes of international diplomacy in the post-Cold War period. The adoption of a treaty through such a framework does not of itself imply that objectives and targets will be met. However, the moral, financial and political pressures that such treaties can bring to bear may be large. Also noteworthy is the way in which the UN environmental strategy has attempted to link issues of environmental protection, environmental sustainability and economic development (the latter particularly in the poorer nations). We came across an example of this linkage in Chapter 2 in our discussions of sustainable development and the Millennium goals. Initiatives through the United Nations are not the only, or even the most important, framework within which international environmental cooperation has taken place. Much of what is important has been dealt with at regional or bilateral levels, and takes place in relatively loose, informal ways.

Why is there a need for international treaties at all? The answer to this question has already been sketched out in the Introduction. Political sovereignty resides principally in nation states. And as the epigraph to this chapter suggests, that state of affairs is likely to remain so for the foreseeable future. There is one important exception to this statement. Countries of the European Union have moved some way towards creating a supranational political institution, although it remains a moot point as to how much substantial sovereign power member states have relinquished to the European Union.

Many environmental impacts of economic activity do not respect national boundaries. As the scale and pervasiveness of these activities increases, so the proportion of activity that has international (rather than intra-national) consequences rises, and also becomes more evident. In the absence of a formal supranational political apparatus with decision-making sovereignty, the coordination of behaviour across

[14] We shall later challenge the view that the number of IEAs is a good indicator of the extent of international environmental cooperation. But we leave this aside for the moment.

[15] Details of IEAs can be found in various sources, many of which are online. The authors have constructed a web page listing the more important IEAs, and providing web links to information about each. The web page is labelled *Treaties.htm* and can be found on the Companion Website. Additionally, this web page points readers to other online sources of information relating to environmental treaties.

countries seeking environmental improvements must take place through other forms of international co-operation. Formal international treaties represent the most visible outcome of that cooperation.

9.2.2 How effective are international environmental agreements?

Looking at the vast number of IEAs that currently are in place, one might suppose that the route of voluntary international environmental agreements has proven to be a particularly fruitful one as far as reaching cooperative outcomes is concerned. But one needs to be cautious before reaching such a conclusion. To answer this kind of question, one needs to look at how effective particular agreements have been in moving towards cooperative outcomes.

Undertaking such an exercise is a daunting task, as it requires the construction of a counter-factual – what would have happened anyway if the IEA had not been reached, so that abatements achieved under an IEA can be compared with the counter-factual estimated abatement levels in the absence of the treaty. In other words, the test of effectiveness of agreements is by comparison of the Nash (non-cooperative) and cooperative outcomes. By this criterion, the literature on IEAs suggests that they are likely to be very limited in their effectiveness.

In his discussion of this issue, Finus (2002) refers to two econometric studies by Murdoch and Sandler (1997a, 1997b) that considered agreed sulphur reduction under the Helsinki Protocol and agreed CFC-reductions under the Montreal Protocol. The authors concluded that the reductions, whilst being large, appear to be more in line with non-cooperative than with cooperative behaviour of governments.[16] For this and similar reasons, Finus also notes that the success of a treaty cannot be inferred from a high participation rate nor from a high degree of compliance.

The research literature on IEAs suggests that performance has been very mixed. Three assertions about the effectiveness of IEAs seem to emerge from the theoretical literature, all of which imply somewhat pessimistic results:

- Treaties tend to codify actions that nations were already taking. Or, put another way, they largely reflect what countries would have done anyway, and so offer little net improvement.
- When the number of affected countries is very large, treaties can achieve very little, no matter how many signatories there are.
- Cooperation can be hardest to obtain when it is needed the most.

Barrett (1990b, 1994a and 1994b) gives many illustrations of these results. The notions that where abatement targets are very undemanding or are little different from what countries would have done anyway, high participation and compliance do not imply treaty effectiveness, and that treaties tend to codify actions that nations were already doing suggest that a treaty with large numbers of signatories – such as the Convention on Biological Diversity which at 2009 had been ratified by 190 states plus the European Union countries – is limited in its capacity to deliver economic benefits. He compares this case with that of the Antarctic Treaty which as of 2009 had just 28 ‘consultative member’ countries.

We shall have more to say about the validity of these, and a series of related, assertions later in this chapter. To set the scene – and to provide an agenda of issues for analysis in later sections – we present, in Box 9.1, a set of ‘stylised facts’ about international environmental cooperation.[17] There is now a huge literature on the economics and politics of international environmental agreements. The stylised facts listed in the box have been extracted from conclusions that have been found with some regularity in that literature. Nevertheless, you should treat these as hypotheses rather than facts, and examine them in the light of the evidence given – and the theoretical explanations offered – in the chapter.

[16] Finus provides some other relevant information here. With regard to the Helsinki Protocol, some members (and even some non-members) had already achieved the reduction target in 1985 when the treaty was signed. Moreover, not only members but also most non-members met or exceeded the 30% sulphur reduction target in 1993.

[17] These ‘stylised facts’ are largely derived from theoretical reasoning rather than observed empirical regularities, so that term may be a misnomer.

Box 9.1 Conditions conducive to effective cooperation between nations in dealing with international environmental problems

Game theory suggests that some conditions are conducive to effective international cooperation. These conditions include the following (each of which will be illustrated in this chapter):

- The existence of an international political institution with the authority and power to construct, administer and (if possible) enforce a collective agreement.
- The output of the international agreement would yield private rather than public goods.
- A large proportion of nation-specific or localised benefits relative to transnational benefits coming from the actions of participating countries.
- A small number of cooperating countries.
- Relatively high cultural similarity among the affected or negotiating parties.
- A substantial concentration of interests among the adversely affected parties.
- The adoption of a leadership role by one 'important' nation.
- A small degree of uncertainty about the costs and benefits associated with resolving the problem.
- The agreement is self-enforcing.
- There is a continuous relationship between the parties.
- The existence of linked benefits.
- The short-run cost of implementation is low, and so current sacrifice is small.
- The time profile of benefits is such that a high proportion of the available benefits are obtained currently and in the near future.
- The costs of bargaining are small relative to the gains expected from cooperation.

9.2.3 The difficulties involved in reaching effective international environmental agreements

We have seen that the absence of an international agency that can enforce binding environmental policy instruments implies that any agreement reached amongst the negotiating countries cannot be enforced by any external agency. Any country entering into such an agreement, and remaining in it thereafter, has to do so voluntarily. International environmental agreements are, therefore, voluntary cooperative activities. Finus (2002, 2004), in a series of excellent review papers, has noted that cooperation faces three fundamental constraints:

1. IEAs have to yield positive net benefit for all potential participants.
2. The parties must agree on the particular design of an IEA by consensus.
3. The treaty must be enforced by the parties themselves.

These are very demanding requirements. Consider first the condition that all potential parties to an IEA must find it beneficial to participate. Although cooperation raises collective welfare, it usually does not lead to an actual Pareto improvement. Without appropriate compensation mechanisms, some countries are likely to be worse off, which will reduce the extensiveness of treaty participation. The Kyoto Protocol is a case in point, as we discuss in detail later in this chapter. In the absence of substantial side-payments, which could offset their perceived costs in terms of growth opportunities, most developing nations are not signatories to the agreement. Most commentators would argue that private benefit–cost calculations led to the USA's failure to ratify the Protocol, and to non-participation in commitments to reduce emissions by developing countries. Moreover, Finus argues that individual countries are more likely to suffer negative net benefits the more ambitious and efficiency-driven are the abatement targets and the more heterogeneous are countries' welfare functions.

Consensus is always a difficult thing to achieve. Finus notes that there may be many ways of designing an IEA that is of positive net benefit to all parties, but finding *one* to which all will agree is often difficult. Negotiation is often held up around matters relating to levels of individual country abatement and their associated abatement burdens, and the magnitudes and distribution of compensation payments.

There are also arguments about the baseline year. As noted in Box 9.1, consensus is hard to achieve when large values are at stake. Finus notes, in this regard, that consensus agreement on declarations of intentions in framework conventions is relatively easy, but that it is far more difficult to achieve in subsequent protocols in which major commitments are at stake. This seems to be exemplified by the United Nations Framework Convention on Climate Change (UNFCCC), and the Kyoto Protocol ratification process.

Consensus also matters because for almost all IEAs, amendments to protocols require unanimity. If no consensus can be reached, the changes are only binding for those participants that accepted the amendments. This partly explains why amendment protocols that tighten emission standards are signed by substantially fewer countries than the original protocols. The condition that the agreement is self-enforcing places major constraints over what is possible. We shall consider this matter further in the next section.

9.2.4 Complete and incomplete environmental agreements

The outcome of a negotiation about an international environmental problem is not restricted to only one of full cooperation or no cooperation at all. A third possibility is partial cooperation: some countries agree to abate pollution (by negotiated amounts), while others behave independently, acting in their own best interests given what the cooperators have agreed upon. This could be described as an incomplete environmental agreement. In this section, we briefly explore how such incomplete cooperation may be an equilibrium outcome. To do this, we use a concept that has been touched on before, but without being fully defined: a self-enforcing international agreement.

Suppose that there are N countries all of which suffer from a common international externality problem and so are potential signatories to an IEA. An agreement is self-enforcing if its terms create incentives on all N parties – cooperators and non-cooperators – to adhere to the agreement once it has come into effect. For this to be the case, the agreement must satisfy the following conditions for each country, $i = 1, \ldots, N$:

1. There is no incentive to renegotiate the agreement.
2. Pay-offs must be such that cheating is deterred.
3. Penalties to countries other than i should not be a disincentive to country i.
4. Penalties to country i should not encourage country i to renegotiate.

Let us think about the kinds of choices that have to be made in arriving at such an agreement. It is usual to proceed on the basis that the first choice to be made by each country is whether or not to participate in an agreement. Secondly, the terms of the agreement must be decided upon. These terms concern how much abatement a signatory will undertake.[18] More precisely, the terms specify a schedule of abatement levels, one for each possible number of other countries acceding to the agreement. Therefore, implicitly or explicitly, the terms include penalties and rewards that reflect what signatories will do if a country were to accede to, or to leave, the group of cooperating countries. This last point is at the heart of how self-enforcing treaties work. Essentially, what happens is that there will be some mechanism whereby if an additional country accedes the signatories increase their abatement (thus rewarding accession), or reduce their abatement if a country leaves (thus punishing defection). This can be stated a little more formally. A self-enforcing international environmental agreement is an equilibrium outcome to a negotiated environmental problem that has the following properties:

- There are N countries in total, of which K choose to cooperate and so $N - K$ do not cooperate.
- Each cooperating country selects an abatement level that maximises the aggregate pay-off of all countries that cooperate.

[18] By signatory here, we mean a 'party' to the treaty in the sense that not only has that country become a signatory to the agreement in question, but also that the decision to participate has been ratified by that country's parliament. This is different from the use of signatory in treaty parlance, where that word does not necessarily entail that the country in question has actually ratified the treaty and so become a 'member' or a 'party'.

- Each defecting (non-cooperating) country pursues its individually-rational unilateral policy.

Choices by each country must also satisfy the two conditions:

- no signatory can gain by unilaterally withdrawing from the agreement;
- no non-signatory can gain by unilaterally acceding to the agreement;

which can be represented by the inequalities $\Pi_s(k^*) \geq \Pi_n(k^* - 1)$ and $\Pi_n(k^*) \geq \Pi_s(k^* + 1)$, where subscripts s and n refer to signatories and non-signatories respectively, and k^* is the equilibrium number of signatory (i.e. participating) countries. The derivation of a solution to this problem is outlined in Appendix 9.1, available on the Companion Website. Here we describe in qualitative terms some of the key results that follow from this kind of framework.[19]

1. Non-signatories and signatories would both do better if *all* cooperate. (In this respect, self-enforcing IEAs resemble a Prisoner's Dilemma game.)
2. But full cooperation is not usually a stable Nash equilibrium (because it is not usually self-enforcing and so is not renegotiation-proof).
3. Non-signatories do better than signatories. (In this respect, the game is like Chicken.)
4. An IEA may enjoy a high degree of cooperation but only if the difference between global net benefits under the full cooperative and non-cooperative solutions is small; when this difference is large, a self-enforcing IEA cannot support a large number of countries.
5. When N is very large, treaties can achieve very little, no matter how many signatories there are.

Barrett (1994a, 1995) was the first to state these (somewhat disheartening) results, and provides the following reasoning and intuition. The larger are the potential gains to cooperation, the greater are the benefits of not participating and so the larger are the incentives to defect. But the larger are the incentives to defect, the smaller will be the number of signatories. The reason here is that when N is large, defection or accession by any country has only a negligible effect on the abatement of the other cooperators. As we shall see later, these results bode badly for attempts to control greenhouse gas emissions. There the gains from cooperation are very large, and so defection is very likely. Given this, it will be difficult to secure agreement among a large number of countries.

Notice that the arguments here presuppose that the product of an environmental improvement is a public good, otherwise it would be impossible for all parties to benefit from increased abatement by any one of them. Then, as with all public goods, free-rider problems potentially arise. Finus points out that we can fruitfully think about the difficulties of achieving successful IEAs in terms of two types of free-riding problem. The first kind arises from the fact that a country may be better off by remaining a non-participant. This is most evident when the number of signatories to an IEA falls short of the total number of countries involved in the externality problem. The second kind arises when a country may be better off by acceding to an IEA but violating its terms (and so in effect no longer being a party to the agreement). We shall leave it to you as an exercise to relate these two kinds of free-riding to the two inequalities stated above.

Finus (2002) argues that such free-riding is particularly damaging to IEAs with explicit and ambitious abatement targets. He provides the following illustrative examples of the first kind of free-riding. CFC and greenhouse gas externalities affect all (200 or so) countries, but by 2002 only 38 industrialised countries had accepted emission ceilings under the Kyoto Protocol, and only 26 countries had signed the Montreal Protocol on ozone-depleting substances by 1987. Although signatories to the Montreal Protocol subsequently rose to 180 parties, the more ambitious amendment protocols counted fewer participants (London, 1990: 153 current parties;

[19] These results have been derived only for some functional specifications and some possible sets of assumptions (including identical countries, and constant marginal benefits of abatement). It is not clear how generally robust the conclusions reported above are to variations in specifications and assumptions. This is a matter of ongoing research.

Copenhagen, 1992: 128; Montreal, 1997: 63; and Beijing, 1999: 11). Despite sulphur being a major air pollutant leading to acid rain, the 1985 Helsinki Protocol counts currently only 22 parties, of which 16 are EU-countries.[20]

As far as the second kind of free-riding is concerned, Finus (2002) cites Keohane (1995, p. 217) as writing '. . . compliance is not very adequate. I believe that every study that has looked hard to compliance [of all major IEAs] has concluded . . . that compliance is spotty.' He also cites the following examples: a study by Brown Weiss and Jacobson (1997, p. 87) that found instances of violations of all IEAs covered by their extensive study; Sand (1997, p. 25), found that over 300 infractions of CITES have been counted per year; and a finding by Heister (1997, p. 68) that all important parties had breached the International Convention for the Regulation of Whaling.

Finally, lest the reader be left with the impression that IEAs have little prospect of real success, two further observations are warranted. First, many important examples of environmental problem affect only small numbers of countries. Where this is the case, cooperative bargaining agreements are relatively easy to obtain. These can often be embodied in *ad hoc* agreements and loose structures. Where the number of countries affected by an environmental problem is large, successful cooperation is harder to achieve. However, these difficulties are lessened if there are large nation-specific gains, and if influential nations are willing to act in the role of leaders. The configuration of pay-offs can be made more conducive to cooperation by linkages between various policy goals (such as debt-for-nature swaps).

Second, it would be wrong to conclude that small IEAs are necessarily inferior to large IEAs. Agreement on ambitious targets might be far easier to obtain between a small group of countries and compliance might be easier to enforce. And as Finus (2002) points out '. . . an inefficient may be superior to an efficient allocation of abatement burdens if it leads to a more symmetric distribution of the gains from cooperation. This may ensure a higher rate of participation and compliance and may put less strain on critical countries so that they agree on higher abatement targets.'

9.3 Other factors conducive to international environmental cooperation

The notion of self-enforcing agreements has proved itself to be a very useful way of thinking about international environmental cooperation. However, as we have seen, it does tend to generate rather pessimistic conclusions about the effectiveness of agreements. It may not be necessary, though, for agreements to be self-enforcing in their own right. In this section we outline some other mechanisms that might be capable of supporting international environmental cooperation.

9.3.1 Role of commitment

A commitment is an unconditional undertaking made by an agent about how it will act in the future, irrespective of what others do. In the context of an IEA, for example, country might commit to a particular level of pollution reduction, irrespective of how many other parties subsequently choose to become members of the IEA.

By giving up the right to change abatement levels in response to changes in the number of parties to an agreement that agree to cooperate and have not subsequently defected (K, in earlier terminology), any agreement that is obtained will not in general be self-enforcing. However, if the commitments are regarded as credible, then – depending on what kinds of commitments are made – it can be possible to achieve and sustain a full (also known as 'complete') IEA. The difficulty here, of course, is that as

[20] Finus (2002) also notes that participation in the framework conventions without specific abatement obligations preceding these protocols is very high (Framework Convention on Climate Change: 186 parties, Vienna Convention (ozone depletion): 180 parties and LRTAP (sulphur emissions): 48 parties). One should also not note that sulphur is a transboundary, but not a global, pollutant and so one might argue that bringing sulphur into the discussion is somewhat misleading.

commitments typically lead to self-sacrifice in some circumstances, it may be hard to make them credible.

One interesting form of commitment is the use of performance bonds.[21] Performance bonds are payments made to an appropriate authority in expectation of compliance with environmental requirements. The bonds are refunded when compliance is achieved. In the context of voluntary agreements between countries, suppose that each party to an agreement negotiated a cooperative solution. Such an agreement would in general be unstable as there is a strong incentive operating on each player to attempt to obtain the benefits of free-riding on the other's pollution abatement. However, suppose that countries made payments (performance bonds) to an independent authority, such as an agency of the United Nations Organisation, at least as large as the net benefit that each would obtain by reneging on the agreement, knowing that such a payment would be refunded only if it did subsequently comply with the agreement. This commitment should be sufficient to make that agreement self-enforcing.

9.3.2 Transfers and side-payments

Suppose that a self-enforcing IEA is only capable of supporting a small number of signatories, *K*. Now imagine that the signatories offer side-payments to induce non-signatories to enter. If these side-payments are larger than the inducements in the original IEA, others will join in cooperation. In some circumstances, such side-payments can bring about a complete IEA, and so maximise collective benefits. However, as the resulting agreement will, by construction, not be self-enforcing, we have the same difficulty as mentioned in the previous sub-section. Such IEAs may not be credible. It seems that side-payment systems will require that signatories find a way to make a credible commitment to the system (and in effect suspend the self-enforcing constraints).

9.3.3 Linkage benefits and costs and reciprocity

It may be possible to secure greater cooperation than the analysis to date has indicated if other benefits are brought into consideration jointly. Doing this alters the pay-off matrix to the game. To see what might be involved here, we note that countries typically cooperate (or at least try to do so) over many things: international trade restrictions, anti-terrorism measures, health and safety standards, and so on. There may be economies of scope available by linking these various goals. Moreover, reputations for willingness to act in the common interest in any one of these dimensions may secure benefits in negotiations about another. What policy makers might try and obtain is linkages over two or more policy objectives so that the set of agreements about these objectives creates overall positive net benefits for the entire set of participants, and net gains which are distributed so that every participant perceives a net linkage gain. In these cases, there can be very substantial gains from international cooperation.

Of course, it must also be recognised that there may be 'additional' costs of cooperation too. These include transaction and enforcement costs, and perceived costs of interdependency itself (such as feelings about loss of sovereignty). The larger are these costs, the smaller are the possible net gains from cooperation. One might also note that the present configuration of international institutions is less than ideal as a facilitator of reciprocal policy linkages. Institutions are often established with a relatively narrowly defined terms of reference (such as WTO: international trade promotion and regulation; IMF and World Bank: responsibility for international financial support and coordination; and agencies of the United Nations Organisation, with domains covering development, the environment, education and cultural affairs, and so on). Attempts have been made from time to time to integrate and link the terms of reference of such institutions, but with at best modest success.[22]

[21] Performance bonds are also known as 'deposit refund schemes' and 'environmental assurance bonding systems'. They have been widely advocated as a general method of implementing the precautionary principle in the context of uncertainty about environmental outcomes. See, for example, Costanza and Cornwell (1992). The recommendations for further readings at the end of this chapter point readers to other useful sources. Performance bonds were also discussed in Chapter 6 and will be again in Chapter 13.

[22] An often-discussed case has been the process of trying to broaden WTO rules of 'fair trade' to include differential standards of environmental protection. There will be more on this matter in Chapter 10. And of continuing discussion is the question of whether the institutional separation of UNDP from UNEP is desirable.

B's strategy / A's strategy	*Defect*	*Cooperate*
Defect	P, P	T, S
Cooperate	S, T	R, R

Figure 9.11 A one-shot Prisoner's Dilemma game

9.3.4 Repeated games

Another mechanism that may enhance the extent of cooperation is repeated interaction between nations. Thus far in this chapter we have been assuming that choices are being made just once. But most environmental problems are long-lasting and require that decisions be made repeatedly. To examine how this may alter outcomes, let us look first at Figure 9.11 which represents the pay-offs in a one-shot game. Here we suppose that the pay-offs have the ranking $T > R > P > S$ and that $S + T < 2R$. The dominant strategy for each player in this game is P.

Now imagine this game being played twice (in two consecutive periods, let us say). The pay-off matrix for this two-shot Prisoner's Dilemma game, viewed from the first of the two periods, is shown in Figure 9.12. Once again, the dominant strategy is P. In fact, this result is true for any fixed and known number of repetitions. However, empirical evidence and experimental games both suggest that cooperation occurs more commonly than theory predicts. What lies behind this? First, cooperation is more likely when communication is allowed. Second, the likelihood of cooperation increases greatly if interaction is never-ending, or its end point is unknown.

A large literature exists that analyses games played repeatedly. We cannot survey that literature here. Suffice it to say that among the many strategies that are possible, some variety of tit-for-tat strategy seems to be sensible. A tit-for-tat strategy usually refers to an iterated (or repeated) prisoner's dilemma game in which (in a two player game) a player cooperates on the first move, and thereafter copies the previous move of the other player. Tit-for-tat strategies tend to encourage cooperation.

B's strategy / A's strategy	*Defect*	*Cooperate*
Defect	$2P$, $2P$	$T + P$, $S + P$
Cooperate	$S + P$, $T + P$	$R + P$, $R + P$

Figure 9.12 The two-shot Prisoner's Dilemma game

However, some results reminiscent of those we have found earlier also emerge. In particular, as N becomes large, cooperation tends to be more difficult to sustain. Indeed Barrett (1994a) shows that even in infinitely repeated games his previous conclusions remain true: an IEA will only be able to support a large number of signatories when gains to cooperation are small; and when the gains are large a self-enforcing IEA can sustain only a smaller number of signatories. Once again, some form of commitment seems to be required if large gains are to be obtained.

9.3.5 Are players only concerned with the returns that they get?

At the start of our discussion of game-theoretic analysis of international environmental problems, we noted that game theory assumes that players are concerned only with the returns that they get. We also remarked that this is a very strong assumption, and one that is not uncontroversial. Many aspects of observed behaviour do not fit well with the notion that motivation is purely driven by 'self-interest rationality'.

With regard to individual persons, experimental evidence strongly shows that they are also much concerned with notions of fairness. The classic experiments are where one makes an offer for a division of money which the other must accept or reject. Both parties behave 'irrationally'. There are suggestions that this is what we might expect from evolutionary psychology given that humans are social animals.

There may, of course, be reasons why concern for fairness does not carry over to nation states. But with regard to climate change, for example, much of the discussion and text of agreements involves arguments about responsibility and fairness. Perhaps this is just 'talk'. But this does not really wash. One cannot fully understand why Annex 1 countries ratified Kyoto without admitting some concern for fairness, for want of a better term, on their part. Certainly the positions taken by many NGOs involve such motivations, and national governments are not immune to their influence.

9.4 Stratospheric ozone depletion

We now turn our attention to two important international environmental problems. The first of these – depletion of ozone concentration in the upper layers of the atmosphere – was of major concern to policy makers in the last two decades of the twentieth century. The fact that it is now much less widely discussed is testament to the relatively high degree of success that has been achieved in ameliorating this problem. Indeed, many commentators regard protection of the ozone layers as being one of the few outstandingly effective international environmental agreements. It is for this reason that we have chosen to include this as our first case study.

Our second case study deals with what is generally regarded as the most important environmental problem faced by humans today, global climate change. Attempts to foster a cooperative approach to avoiding excessive climate change have dominated much of the research agenda in the last decade, and have spawned a large number of innovative institutional mechanisms and policy initiatives. For these reasons, much of the rest of this chapter will deal with the science and economics of global climate change. But first, we consider the issue of ozone depletion.

9.4.1 Summary of problem

Ozone is produced in the upper layers of the atmosphere by the action of ultraviolet light on oxygen molecules. The processes determining the concentrations of upper-atmospheric ozone are complex and incompletely understood. It is known that ozone concentration is in a constant state of flux, resulting from the interaction of decay and creation processes. Several naturally occurring catalysts act to speed up natural rates of decay; these catalysts include oxides of bromine, chlorine, nitrogen and hydrogen. There are large, naturally caused variations in these concentrations by time, spatial location and altitude. For example, normal dynamic fluctuations in ozone concentrations are as large as 30% from day to night, and 10% from day to day (Kemp, 1990).

During the early 1970s, scientific claims that ozone was being depleted in the stratosphere were first made. These original claims were not satisfactorily verified, but in the mid-1980s the discovery of the so-called hole in the ozone layer over Antarctica led the scientific community to conclude that serious reductions in mean atmospheric ozone concentrations were taking place, with additional large reductions in particular spatial locations. The downward trend in ozone concentrations was attributed to inadvertent human interference with the chemistry of the atmosphere, related to prevailing patterns of air pollution. Over the continent of Antarctica, the fall in concentration (relative to its 1975 level) was estimated to be in the interval 60–95%, depending upon the place of measurement (Everest, 1988).

Although much progress has been made towards understanding the chemistry of ozone depletion, we remain uncertain even as to the recent historical rates of depletion. Estimates of the actual rates of depletion experienced have been considerably lowered since the initial studies were published, and forecast depletion rates are now much less than early predictions. Current models forecast depletion to be no more than 5% on average over the next 50 years, as compared with initial predictions of depletions of up to 20%.

There are several ways in which human impacts on the ozone layer take place. Two of these – nuclear radiation and aircraft emissions – appear to have relatively little effect at present, but are potentially important. Evidence also implicates a number of other chemicals as ozone depleters, in particular nitrous oxide (associated with traffic and agricultural activity), carbon tetrachloride, chloroform, and trichloroethane.

The dominant anthropogenic proximate cause of ozone depletion appears to be halogen atoms (chlorine and bromine) in the stratosphere, produced by photodissociation of chlorofluorocarbon (CFC) compounds, known as freons, and of bromofluorocarbon compounds, known as halons. These substances are transported into the stratosphere after being emitted at the surface. They then act as catalysts to the decay of ozone, adding to the effects of the natural catalysts we mentioned earlier, and growing in effect rapidly as emissions of CFCs and halons increased.

Many forms of CFC exist, two of them – CFC-11 and CFC-12 – being the dominant forms. The most important sources of CFC emissions by quantity are the production, use and disposal of aerosol propellants, cushioning foams, cleaning materials and refrigerative materials. In some cases, such as in aerosol uses, the release of the gas occurs at the time of manufacture or within a relatively short lapse of time after manufacture. In other cases, the release can occur at much later dates as items of hardware such as refrigerators and air-conditioning units are scrapped. Estimates by Quinn (1986) suggested that CFCs have very high income elasticities of demand. Hence, if CFC gases were not subject to control, their use would rise very rapidly as world incomes increase. Until recently, the North has been far more important than the South in terms of the quantities of emissions of ozone-depleting substances. However, this will soon change as economies of the South undergo rapid economic growth.

What would be the effects of a continuing depletion of the atmospheric ozone layer? The consequences follow from the fact that ozone plays a natural, equilibrium-maintaining role in the stratosphere through

(a) absorption of ultraviolet (UV) radiation, and
(b) absorption of infrared (IR) radiation.

The absorption of IR radiation implies that CFC substances are greenhouse gases, contributing to global climate change. This aspect of ozone depletion is discussed in the following section. Here we focus on the role played by halons and CFCs in depleting the concentration of ozone in the upper atmosphere and leading to increased UV radiative flows. The ozone layer protects living organisms from receiving harmful UV radiation. It is now virtually certain that ozone depletion has increased the incidence of skin cancer among humans. Connor (1993) estimates that a 1% depletion in ozone concentration would increase non-malignant skin cancers by more than 3%, but by rather less for malignant melanomas. An early study by the United States Environmental Protection Agency estimated that human-induced changes in the ozone layer would cause an additional 39 million cases of skin cancer during the next century, leading to 800 000 additional deaths (Kemp, 1990).

Effects which may occur, but about which much doubt remains, include effects on human immune systems (including activation of the AIDS virus), radiation blindness and cataract formation, genetic damage to plants and animals, and losses to crops and other plant or animal damage. Of particular concern is the apparent damage to marine plankton growth – the importance of plankton in many food chains suggests that this may become a critical issue during the next century. Increased UV radiative flows are also likely to accelerate the degradation of polymer plastic materials.

Table 9.4 Costs and benefits of CFC control in the United States

Level of control	Discounted benefits ($ billion)	Discounted costs ($ billion)
80% cut	3533	22
50% cut	3488	13
20% cut	3396	12
Freeze	3314	7

Source: Adapted from EPA (1989)

Some indication of the likely magnitudes of the costs and benefits of control was given in a United States EPA 1989 study, the results of which are reported in Table 9.4. It is sometimes thought that rapidly rising marginal costs of abatement mean that very large proportionate reductions in pollutant emissions would be prohibitively expensive. If the estimates in Table 9.4 are trustworthy that is not true in the case of CFC reduction. It is clear from Table 9.4 that the costs of CFC reduction do rise as the magnitude of abatement is increased, but not in a sharply increasing manner. Moreover, the benefits rise substantially as abatement is increased. Given these numbers, almost complete elimination of the pollutant emissions is economically warranted.

9.4.2 Action to date on abating emissions of ozone-depleting substances

Lengthy spans of time can elapse between first identifying the need for collective action and the realisation of that action. Ozone layer protection was first discussed at a meeting of the United Nations Environmental Program in 1976, but preparation for a treaty did not start until 1981. The Vienna Framework Convention was adopted in 1985 at which agreements were made for international cooperation in research, monitoring and the exchange of information. It is generally agreed that preliminary agreements of this form are of great importance. The acquisition of information about the costs and benefits of an environmental control programme reduces uncertainty and improves the chances of an effective response. For global problems, such as ozone-layer depletion, the accumulation of scientific evidence seems to be required before nations perceive the need to act. The discovery of the hole in the ozone layer in 1985, and publications in 1987 and 1989 of research by the US EPA were also pivotal in generating international support for control of ozone-depleting substances.

By 1989, 27 countries had ratified the Vienna Convention. The first targets for reduction in emissions of ozone-depleting substances were agreed upon in the Montreal Protocol in 1987, which entered into force in 1989. The Protocol parties (at that time 24 mainly industrialised countries) agreed to phased reductions in domestic consumption and production of ozone-depleting substances, and in particular to cease the production of chlorofluorocarbons (CFCs) by 1996. By 1995, at which time amendments were made to the treaty, 149 countries had ratified the Protocol. Developing countries could increase CFC production until 1999, after which it must be progressively reduced until CFC production was to cease in 2010. The London Protocol, signed in July 1990 by 59 nations, agreed to a complete phasing out of halons and CFCs by the year 2000. In addition, controls were agreed on two other substances implicated in the depletion of ozone, carbon tetrachloride (to be eliminated by 2000) and methyl chloroform (by 2005). Financial support was made available to assist in the funding of projects to substitute from ozone-depleting substances in poorer countries.

International action to control the production and use of CFCs is widely regarded as the outstanding success of international environmental diplomacy. The agreements led to a rapid decline in global CFC emissions, although most of this was achieved in the developed countries (with the United States adopting a tradable permit scheme for domestic CFC usage). The emissions of ozone depleting substances (ODS) controlled under the Montreal Protocol have declined significantly since the 1990s. By 2004 the emissions of these gases were about 20% of their 1990 level. It is expected that ozone concentrations in the stratosphere will gradually return towards pre-industrial levels by the middle to late part of this century. Nevertheless, during this process of return to an earlier steady state, ultraviolet levels are expected to rise by a further 10–15%, with a comparable increase in the incidence of skin cancers.

Several factors mentioned in Box 9.1 have been important contributors in ozone depletion control. First, there has been a high concentration among the sources of CFC emissions. For example, in 1986 just three countries – the Soviet Union, the USA and Japan – accounted for 46% of global CFC emissions. A high concentration of interests implies that there are relatively few parties to the problem. This implies, in turn, that those countries attempting to negotiate an agreement need have little concern about free-riding among treaty non-participants, and so are more willing to become signatories.

Second, we see here the importance of one influential nation being willing to adopt a leadership role, particularly where that country has a large beneficial stake in the outcome of negotiations. The USA was a major driving force in discussions that led to the Montreal and subsequent protocols.[23] As we shall see later, the absence of any single major country willing to act in this way constitutes a serious obstacle to progress on control of global climate change.

Barrett has argued that the success of the Montreal Protocol in facilitating the setting of stringent targets for emissions of ozone-depleting substances, against a background of scientific uncertainty, derives from the fact that the Protocol reversed the incentives for ‘free-riding’ and generated self-enforcing dynamics. The Protocol’s incentive structures generated both

[23] It has been suggested that the enthusiasm of large United States CFC-producing companies for compulsory control of ozone depleting substances arises from the fact that US company-owned patents on CFC emitting substances were expiring in the period just prior to the Montreal Protocol. The potential profitability streams from newly patented CFC and halon substitutes far exceeded those that would have been generated by continued production under post-patent generic CFC-emitting substances.

'sticks' (for example, trade restrictions between signatories and non-signatories, including a ban on trade in products containing these ozone-depleting substances such as refrigerators) and 'carrots' (in the form of side-payments to pay to developing countries some of the costs of introducing chlorofluorocarbons (CFC)-free technologies). These incentives aligned every country's self-interest with participation in the IEA. Trade restrictions boosted market size for CFC substitutes, speeding-up the control mechanisms. Finally, Barrett points to the presence of some other of the favourable conditions listed in Box 9.1, notably low treaty-implementation costs; low costs (yet high profitability) of developing CFC and halon substitutes; and large private benefits of complying (principally in the form of avoided skin cancers and other consequences).

A continuing decline in ozone-layer depletion will depend upon the developing countries substituting away from CFCs as industrial output rises. At the end of the millennium, CFC production was increasing very rapidly in these economies. However, initial very pessimistic forecasts of growth in CFC and halon emissions by developing economies have turned out to be largely misplaced, and China has recently committed to eliminate CFC emissions completely by 2010.

9.5 Global climate change

No discussion of global climate change can be complete without consideration being given to the work carried out by, and the reports of, the Intergovernmental Panel on Climate Change (IPCC). The IPCC, established by the United Nations Environment Programme (UNEP) and the World Meteorological Organization (WMO), has the task of providing a clear scientific view on the current state of climate change and its potential environmental and socio-economic consequences. IPCC is an intergovernmental body, open to all member countries of UN and WMO.

The Panel itself, comprised of government delegations of all member countries, meets approximately once a year at the plenary level. These Sessions are attended by officials and experts from national ministries, agencies and research institutions, and from participating organisations. Major structural and procedural decisions and the structure and mandate of IPCC Working Groups and Task Forces are taken by the Panel in plenary session. The Panel decides also on the scope and outline of IPCC reports.

The IPCC does not itself carry out original research, nor does it monitor climate-related data or parameters. Rather, it reviews and assesses the most recent scientific, technical and socio-economic information produced worldwide relevant to the understanding of climate change. The bulk of this work is carried out by thousands of scientists from all over the world who contribute on a voluntary basis as authors, contributors and reviewers. These reviews and assessments are expected to reflect differing viewpoints existing within the scientific community. The work of the IPCC, and its periodic reports, are designed to be policy relevant, while being policy neutral and never policy prescriptive.

The IPCC is currently organised in three working groups. Working Group I deals with 'The Physical Science Basis of Climate Change', Working Group II with 'Climate Change Impacts, Adaptation and Vulnerability' and Working Group III with 'Mitigation of Climate Change'. The IPCC has also a Task Force on National Greenhouse Gas Inventories, the main objective of which is to develop and refine a methodology for the calculation and reporting of national GHG emissions and removals.

To date (2009), four 'Assessment Reports' have been produced by the IPCC. The first was published in 1990, and played a decisive role in leading to the creation of the United Nations Framework Convention on Climate Change (UNFCCC), the key international treaty to reduce global warming and cope with the consequences of climate change. The Second Assessment Report was published in 1995, providing a key input into adoption of the Kyoto Protocol in 1997. The Third Assessment Report was published in 2001, and the Fourth in 2007.

The discussion of global climate change in this chapter of the textbook focuses on policy aspects. However, we begin with a brief summary of the underlying science, which draws very heavily on the latest, fourth, assessment report (AR4) of the Intergovernmental Panel on Climate Change (IPCC), which became available in 2007.[24] The reader who wishes to learn more about the underlying science can find good accounts in the references given at the end of this chapter.

9.5.1 What determines Earth's climate?

Climate is 'long-term average weather', and is usually taken to cover the mean and variability of temperature, precipitation and wind over a period of time. It is determined by the characteristics and evolution of a system which comprises the atmosphere, land surface, snow and ice, oceans and other bodies of water, and living things. This 'climate system' evolves over time under the influence of its own internal dynamics and changes in external factors or 'forcings' that affect climate. Forcings are of two kinds: natural phenomena such as volcanic eruptions and solar variations; and human-induced changes in atmospheric composition.

Incoming shortwave solar radiation powers the climate system. Changes in the radiation balance of the Earth (known as 'Radiative Forcing' and explained below in Box 9.2) can be altered in three fundamental ways:

1. changes in the incoming solar radiation (e.g. by changes in Earth's solar orbit or in the internal activity of the Sun);
2. changes in 'albedo', the fraction of solar radiation that is reflected (e.g. by changes in cloud cover, atmospheric particles or vegetation); and
3. alterations in the transmission of longwave radiation from Earth back towards space (e.g., by changing greenhouse gas concentrations).

Climate responds directly to such changes, as well as indirectly, through a variety of feedback mechanisms.

In accordance with principles of thermodynamics, there must be balance between the amounts of incoming and outgoing radiation. Over the long term, the amount of incoming solar radiation absorbed by the Earth and its atmosphere is balanced by the release to space of an equal amount of outgoing longwave radiation. This is implicit in the energy fluxes listed at the top of Figure 9.13, which show that net incoming solar radiation ($342 - 107 = 235\ Wm^{-2}$) is balanced by $235\ Wm^{-2}$ of outgoing longwave radiation.

Figure 9.13 repays further examination by the insights it brings to understanding of climate and the factors changing climate. The amount of energy reaching the top of the Earth's atmosphere each second, averaged over the entire planet, is approximately 342 Watts per square metre (Wm^{-2}). About 30% of the solar radiation that reaches the top of the atmosphere is reflected back to space. Roughly two-thirds of this reflectivity is due to clouds, atmospheric gases, and aerosols.[25] Light-coloured areas of the Earth's surface – mainly snow, ice and deserts – reflect the remaining one-third of the sunlight.[26]

The incoming solar radiation energy that is *not* reflected back to space is absorbed by the Earth's surface and atmosphere, approximately 235 Watts per square metre (Wm^{-2}).[27] To balance the incoming energy, the Earth itself must radiate, on average, the same amount of energy back to space. The Earth does this by continuously emitting outgoing longwave, infrared radiation. The warmer an object, the more heat energy it radiates. To emit $235\ Wm^{-2}$, a temperature of around $-19°C$ is required. This is much colder than the conditions that exist at the Earth's surface. So how is this necessary balance maintained?

The reason the Earth's surface is kept much warmer than this (at an average surface temperature of about 14°C) is the presence of greenhouse gases,

[24] The material in the sections that summarise the underlying science are covered in, and draw heavily on, the Report of Working Group I of the IPCC, 'Climate Change 2007: The Physical Science Basis' (IPCC, 2007a). Accompanying this is an excellent 'Frequently Asked Questions' appendix. The material presented there is the basis for what is written in Section 9.5.1, a large part of which uses words taken directly from that document.

[25] Aerosols are small particles in the atmosphere, some of which are naturally produced, such as those that result from volcanic explosions, but others of which are human-induced.

[26] To ensure that you can relate the text to the contents of Figure 9.13, note that $107\ Wm^{-2}$ of the $342\ Wm^{-2}$ incoming solar radiation (i.e. just over 30%) is reflected back into space. Of this $107\ Wm^{-2}$, $30\ Wm^{-2}$ (roughly one-third) is reflected from the earth's surface by albedo affects, and $77\ Wm^{-2}$ (the remaining two-thirds) is reflected back by clouds, atmospheric gases and aerosols.

[27] That is, 168 (absorbed by Earth's surface) plus 67 (absorbed by atmosphere).

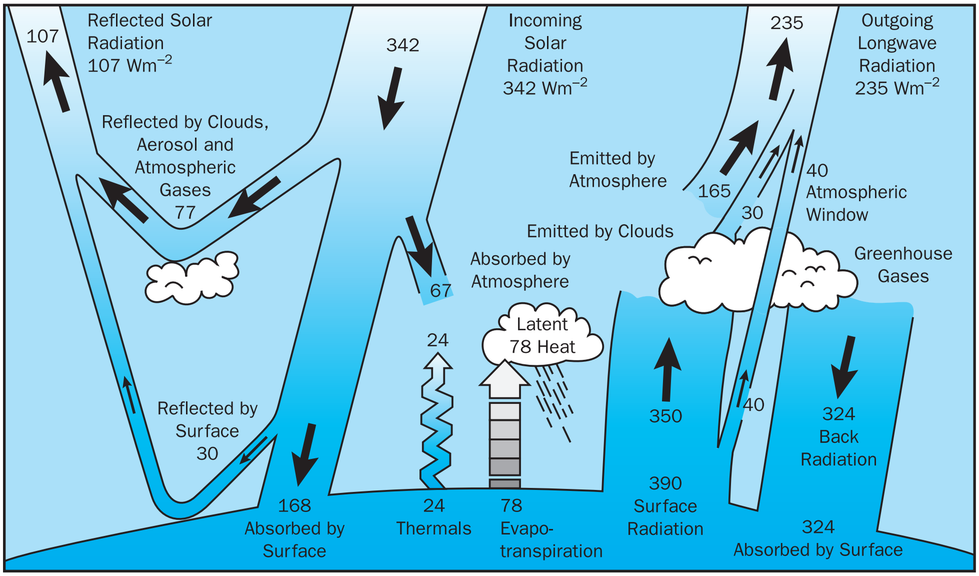


Figure 9.13 Estimate of the Earth's annual and global mean energy balance.
Over the long term, the amount of incoming solar radiation absorbed by the Earth and atmosphere is balanced by the Earth and atmosphere releasing the same amount of outgoing longwave radiation. About half of the incoming solar radiation is absorbed by the Earth's surface. This energy is transferred to the atmosphere by warming the air in contact with the surface (thermals), by evapotranspiration and by longwave radiation that is absorbed by clouds and greenhouse gases. The atmosphere in turn radiates longwave energy back to Earth as well as out to space.
Source: IPCC AR4, Working Group 1 Main Report, FAQ 1.1, Figure 1. Available online at http://www.ipcc.ch/pdf/assessment-report/ar4/wg1/ar4-wg1-faqs.pdf. The IPCC document itself gives as its source for this diagram Kiehl and Trenberth (1997)

which act as a partial blanket for the longwave radiation coming from the surface. Greenhouse gases absorb thermal infrared radiation emitted by the Earth's surface, by the atmosphere itself, and by clouds. These components in turn emit atmospheric radiation to all sides, including downward to the Earth's surface. Thus, greenhouse gases trap heat within the surface-troposphere system.[28] Thermal infrared radiation in the troposphere is strongly coupled to the temperature of the atmosphere at the altitude at which it is emitted. In the troposphere, the temperature generally decreases with height. Effectively, infrared radiation emitted to space originates from an altitude at which the average temperature is −19°C, and thereby balances the net incoming solar radiation. The Earth's surface, though, is kept at the much higher average temperature of +14°C, allowing current ecological systems to be maintained. This blanketing is known as the *natural greenhouse effect*.

The most important greenhouse gases (GHGs) in terms of temperature forcing impacts are water vapour and carbon dioxide. But these are not the only significant GHGs. Agricultural activity, and the decomposition and disposal of waste, are important emitters of methane, another GHG. Chlorofluorocarbons (CFC), halons, nitrous oxide, and ozone emissions also act as global-warming substances. Clouds also exert a blanketing effect similar to that of the GHGs; however, this effect is offset by their reflectivity, such that in net terms and on average, clouds tend to have a cooling effect on climate.

Most importantly for what concerns us in this chapter is that (as we shall show shortly) human activities have led, and continue to lead, to the release of additional quantities of GHGs.[29] The

[28] The troposphere is the lower layer of the Earth's atmosphere.

[29] Additional in the sense of being greater than is required to sustain a surface mean temperature close to that which has existed over recent historical times.

cumulative effect of these additional GHG releases is an increased infrared opacity of the atmosphere, leading to a strengthening of the 'blanketing effect' described above and so to additional radiative forcing that intensifies the magnitude of the greenhouse effect. This is the so-called *enhanced greenhouse effect*, which will gradually lead to higher average global surface and near-surface temperatures, and other changes in the Earth's climate.

All of the science relating to climate determination inevitably contains some degree of uncertainty. But for the processes we have described so far in this section, that uncertainty largely relates to the precise parameterisation or quantification of relationships. However, when it comes to predicting climate in the future, uncertainty becomes more profound, as we show at various places in the pages that follow. But one particular source of uncertainty warrants mention at this point: this is the presence of *climate feedback mechanisms*. An interaction between processes in the climate system is called a climate feedback when the result of an initial process triggers changes in a second process that in turn influences the initial one. A positive feedback intensifies the original process, and a negative feedback reduces it. There are several potentially important feedback mechanisms in the climate system that can amplify or diminish the effects of a change in climate forcing. For example, as rising concentrations of greenhouse gases warm Earth's climate, snow and ice begin to melt. This melting reveals darker land and water surfaces that were beneath the snow and ice, and these darker surfaces absorb more of the Sun's heat, causing more warming, which causes more melting, and so on, in a self-reinforcing cycle. This feedback loop, known as the 'ice-albedo feedback', amplifies the initial warming caused by rising levels of greenhouse gases. The current state of knowledge about climate feedbacks is very incomplete, at least in part because they may be complex and non-linear, and are likely to become significant only when global means temperature lies well outside the range for which we have any observable empirical record. Hence, much of our current beliefs about possible magnitudes of climate feedbacks is based on theoretical conjectures. Detecting, understanding and (hopefully) quantifying climate feedbacks is a focus of much ongoing research.

Figure 9.13 also enables us to gain some insight into the drivers of regional variations in climate. Because the Earth is a sphere, more solar energy arrives for a given surface area in the tropics than at higher latitudes, where sunlight strikes the atmosphere at lower angles. Energy is transported from the equatorial areas to higher latitudes via atmospheric and oceanic circulations, including storm systems. Energy is also required to evaporate water from the sea or land surface; this energy – called latent heat – is released when water vapour condenses in clouds. Atmospheric circulation is primarily driven by the release of this latent heat. Atmospheric circulation in turn drives much of the ocean circulation through the action of winds on the surface waters of the ocean, and through changes in the ocean's surface temperature and salinity (through precipitation and evaporation).

Due to the rotation of the Earth, the atmospheric circulation patterns tend to be more east–west than north–south. Embedded in the mid-latitude westerly winds are large-scale weather systems that act to transport heat toward the poles. These weather systems are the familiar migrating low- and high-pressure systems and their associated cold and warm fronts. Because of land–ocean temperature contrasts and obstacles such as mountain ranges and ice sheets, the circulation system's planetary-scale atmospheric waves tend to be geographically anchored by continents and mountains, although their amplitude can change with time. Changes in various aspects of the climate system, such as the size of ice sheets, the type and distribution of vegetation or the temperature of the atmosphere or ocean, will influence the large-scale circulation features of the atmosphere and oceans.

9.5.2 How would GHG emissions and atmospheric concentrations change over the coming century and beyond if no additional controls were imposed?

In order to be in a position to make any sensible policy decisions about responses to global climate change, one needs to have information about how climate is likely to change over the short-, medium- and long-term futures. A necessary (although not a sufficient) condition for predicting key characteristics of the Earth's climate is information about the future time paths of GHG emissions. To see why, let us review what we know so far. Climate change is heavily driven by the atmospheric *concentrations* of the various greenhouse gases. Being a set of stock pollutants, GHG concentrations at any point in time depend on the rates of emissions of those gases at all previous points in time, and on the extent to which various other sinks have sequestered atmospheric GHGs (or the amounts that have decayed into harmless forms) at all previous points in time. How much global climate change will occur over the next century and beyond is, therefore, partly predetermined by the dependence of GHG concentrations on previous net emissions. But it will also depend on the magnitudes of future GHG emissions and actions that affect the size of various carbon and other sinks.

As one cannot know with certainty what those emissions rates will be, forecasts of future emissions are required. However, predictions about GHG emissions over medium- and long-term horizons are very sensitive to the assumptions made by forecasters.[30] Moreover, the uncertainty surrounding predictions errors will tend to increase the further ahead is the time for which predictions are being made. The main assumptions required relate to population and economic growth rates, product innovation and product-mix choices, changes in energy mix (particularly between fossil fuel and non-fossil-fuel energies, and between changes in the proportions of different fossil fuels used), rates of technological progress and energy efficiency improvement, and income and wealth distributions between and within nations. And as net emissions also depend on land-use changes, such as rates of tropical deforestation, forecasts will also need to factor in the various drivers of land-use changes.

The evolution of GHG emissions changes will also depend on policy choices made by governments and their associated control measures. Indeed, even attempting to define a 'Business As Usual' forecast (one in which there are no additional policy measures introduced) is an exercise fraught with difficulty, given commitments already made to abate GHG emissions and the time lags involved in programme implementation.

As a result, the range of plausible predictions is large, even for relatively short forecast horizons. This creates problems in building up a consistent research picture about climate change. In the early days of climate change research, modelling teams tended to work to independent agenda, making the task of comparison of research output rather difficult. More recently, formal and informal coordination of research efforts has led to some 'standardisation' of agenda. Some of this has resulted from the formation of the Intergovernmental Panel on Climate Change (IPCC) in 1988, and the direction this imparted to climate change research. Of particular interest was the IPCC Special Report on Emission Scenarios (SRES, 2000). Four SRES scenario 'families' were developed: A1, A2, B1 and B2. The A1 family contains three distinct scenario 'groups', each of which makes a different assumption about the evolutions of energy technology. Specifically, A1F1 assumes a fossil fuel intensive future; A1B assumes a 'balanced' future; and A1T assumes a predominantly non-fossil fuel future. Table 9.5 describes the characteristics of these six scenario groups. The SRES scenarios were developed for, and first used in, the IPCC's Third Assessment Report (TAR) in 2001, and were also used for the 2007 Fourth Assessment Report (AR4) with updated parameter estimates.

The AR4 Reports do not only use updated SRES scenarios. A large number of additional scenarios have been developed since 2001, collectively known as 'post-SRES' scenarios. In many cases, the post-SRES scenarios differ markedly in nature from the SRES scenarios listed in Table 9.5. They typically do not follow the SRES methodology of constructing scenarios using narratives describing a set of plausible, coherent, but very different, stories about how the world may develop over the next one hundred years or so. Instead, post-SRES scenarios typically take the main drivers of emissions mentioned earlier, and use top-down macroeconomic modelling or bottom-up 'engineering' techniques to develop expected values of those drivers at various points in time, and their associated emissions. Often

[30] The word 'assumption' is being used in a loose sense, and will include estimates from various modelling exercises, best guesses, and personal judgements.

Table 9.5 SRES emission scenarios

Family	Description
A1	Future of very rapid economic growth, global population peaks in the middle of the 21st century and declines thereafter, and rapid introduction of new and more efficient technologies. Underlying themes are regional convergence, capacity-building, increased social and cultural interactions, substantial reduction in regional differences in per capita income. Three scenario 'groups' identified within the A1 family: **A1FI** Fossil-fuel intensive **A1T** Non-fossil energy technology emphasis **A1B** Balanced fossil/non-fossil emphasis
A2	Very heterogeneous world of self-reliance and preservation of local identities. Fertility patterns across regions converge only very slowly, so global population continues to increase. Economic development is primarily regionally oriented. Per capita economic growth and technological progress more fragmented and slower than in other scenarios.
B1	A convergent world. Future of very rapid economic growth, global population peaks in the middle of the 21st century and declines thereafter. Rapid changes in economic structures towards a service- and information-oriented economy. Reductions in material intensity and introduction of cleaner, resource-efficient technologies. Underlying themes are global solutions to sustainability problems, improved equity, but without additional climate initiatives.
B2	World in which emphasis is on local solutions to economic, social and environmental sustainability. Continuously increasing population (but at a rate slower than in A2). Intermediate levels of economic development, less rapid and more diverse technological change than in A1 and B1. Environmental protection and social equity focused on at local and regional levels.

such exercises provide estimates of confidence intervals or more general types of uncertainty bounds around those central estimates.

Emissions forecasts corresponding to SRES and post-SRES scenarios have been produced by a relatively large number of international expert modelling groups. Each scenario generates a 'reference' or baseline projection of GHG emissions, conditional on a set of socio-economic and technological assumptions. There is considerable variety in the scope of emissions modelling by these groups. Some have focused on the path of CO_2 emissions alone, while others have adopted a broader multi-gas approach. Modellers have also made different choices regarding sectoral coverage, some dealing with only the energy supply and industrial sectors, while others have factored in changes in GHG emissions from land-use changes, and some have attempted to provide all-encompassing emissions forecasts. Not surprisingly, this heterogeneity of approach yields substantial divergence in forecasts. But differences in scope of coverage are not the only sources of forecast variability. Emission projections will also differ from one modelling group to another because of variations in model structure, choice of functional forms used in estimating equations and associated parameter values, and in auxiliary assumptions employed by each group.

A second source of difference between SRES and post-SRES emissions scenarios arises from the property that SRES scenarios were explicitly constructed *not* to include additional climate control initiatives and so, for example, assume that Kyoto targets are *not* implemented. This has proven to be problematic since 2001, given that some additional policy-induced GHG abatement has been made since then, and commitments have been made by various governments to build further on those abatement efforts in coming decades. Furthermore, some SRES scenarios are equivalent to outcomes that would follow from additional control measures being implemented, making questionable the validity of a distinction between scenarios that do and do not include additional control measures over the chosen forecast horizons. As a result, several of the later, post SRES, scenarios take the phrase 'additional climate control' to mean additional measures since the time at which the scenario was constructed, and so do include estimated impacts of Kyoto mitigation measures.

Figure 9.14 shows projections of annual global GHG annual emissions up to the year 2100 for the six SRES scenarios and the 80th percentile range of recent post-SRES scenarios. Dashed lines show the full range of post-SRES scenarios. The emissions include CO_2, CH_4, N_2O and F-gases. Notice that the vertical scale in this diagram is measured in *carbon dioxide equivalent units*. The use of 'equivalent carbon dioxide emission' has become a standard metric for comparing emissions of different greenhouse gases, and so allowing for the construction of

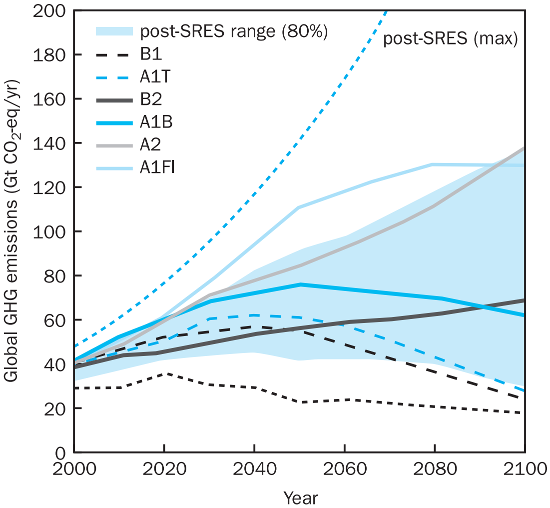


Figure 9.14 Global GHG emissions (in $GtCO_2$-eq per year) in the absence of additional climate policies

The figure shows six illustrative SRES marker scenarios and 80th percentile range of recent scenarios published since SRES (post-SRES) (gray shaded area). Dashed lines show the full range of post-SRES scenarios. The emissions include CO_2, CH_4, N_2O and F-gases.
Source: Figure 3.1, IPCC (2007), AR4 Synthesis Report. Available online at http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4-syr.pdf

meaningful aggregate measures of 'greenhouse gas' emissions. For any individual greenhouse gas, equivalent carbon dioxide (CO_2) emission is defined as the amount of carbon dioxide emission that would cause the same total amount of *radiative forcing*, over a given time horizon, as an emitted amount of that greenhouse gas. The equivalent carbon dioxide emission is obtained by multiplying the emission of a greenhouse gas by its *Global Warming Potential* for the given time horizon. For a mix of greenhouse gases it is obtained by summing the equivalent carbon dioxide emissions of each gas. The terms radiative forcing and global warming potential are explained in Box 9.2.

What is evident from Figure 9.14 is the huge range of emissions projections generated by this exercise, particularly as the time period ahead is extended. By 2100, even ignoring projections outside the 80th percentile range, estimated emissions range between about 25 and 135 $GtCO_2$-eq per year. It is not just the magnitudes that vary but also their profile, with some showing monotonic growth and others showing marked falls in emissions after high-points reached in intermediate years. This is particularly interesting given that all of these scenarios were in principle to assume that no 'additional' climate change measures are adopted along the projected paths.

Although much of the debate about the greenhouse effect is couched in terms of emissions, it is the concentration rate of greenhouse gases in the atmosphere rather than emissions *per se* that drives global mean temperature. So any analysis of the greenhouse effect requires that the predicted path through time of GHG emissions be mapped into the implied atmospheric GHG concentrations. The way in which GHG emissions affect their atmospheric concentrations is apparently simple: emissions add to the stock and decay reduces the stock.[31] For the flow–stock relationship that links GHG emissions to concentrations, two processes are of particular importance.

1. The rate of decay of the GHG stock depends on the 'active' residence time of GHG molecules in the atmosphere. The expected lives of GHG molecules range from a few weeks for tropospheric ozone to 100 years or more for CFCs, with CO_2 towards the upper end of that range. Two consequences follow from this. First, emissions will have a persistent impact on global climate change, given the long-term residency of existing stocks. Second, as the composition of different gases in the overall stock of GHGs alters, so the average lifetime of the composite GHG stock will alter.
2. The net change in GHG stocks in the atmosphere is influenced by the operations of various 'sinks' that sequester CO_2 and other greenhouse gases. It is known, for example, that the oceans absorb some carbon dioxide. But

[31] The relationship between pollution flows and stocks will be analysed in some depth, and in a more general context, in Chapter 16.

Box 9.2 Radiative forcing and Global Warming Potential

Radiative forcing (RF) is a concept used for quantitative comparisons of the strength of different human and natural agents in causing climate change. Technically, RF is measure of the change in *net irradiance* (in units of Watts per square metre) in a given climate system. Net irradiance is the difference between the incoming radiation energy and the outgoing radiation energy in that system due to change in an external driver of climate change, such as a change in the atmospheric CO_2 concentration or presence of aerosols. Radiative forcing is usually quantified as the rate of energy change per unit area of the globe as measured at the tropopause, the boundary between the troposphere (the lowest part of the atmosphere where clouds and weather phenomena occur) and the stratosphere (the region of the atmosphere above the troposphere). It is customary to measure radiative forcing as the change relative to the year 1750.

Intuitively, RF is a measure of how the energy balance of the Earth-atmosphere system is influenced when factors that affect climate are altered. The word radiative arises because these factors change the balance between incoming solar radiation and outgoing infrared radiation within the Earth's atmosphere. This radiative balance controls the Earth's surface temperature. The term forcing is used to indicate that Earth's radiative balance is being pushed away from its normal state. When radiative forcing from a factor or group of factors is evaluated as positive, the energy of the Earth-atmosphere system will ultimately increase, leading to a warming of the system. In contrast, for a negative radiative forcing, the energy will ultimately decrease, leading to a cooling of the system.

Our interest in this chapter lies primarily in anthropogenically induced positive changes in RF. It is these that generate the so-called 'enhanced greenhouse effect'. The combined anthropogenic RF is estimated to be +1.6, with a 95% confidence interval of [+0.6, +2.4]. Thus human-induced RF is strongly and significantly positive. IPCC 2007 states that 'This RF estimate is *likely* to be at least five times greater than that due to solar irradiance changes'. Moreover, IPCC reports that for 'the period 1950 to 2005, it is *exceptionally unlikely* that the combined natural RF (solar irradiance plus volcanic aerosol) has had a warming influence comparable to that of the combined anthropogenic RF'.

Global Warming Potential (GWP) is an index, based upon radiative properties of well-mixed greenhouse gases, measuring the radiative forcing of a unit mass of a given well-mixed greenhouse gas in the present-day atmosphere integrated over a chosen time horizon, relative to that of carbon dioxide. The GWP represents the combined effect of the differing times these gases remain in the atmosphere and their relative effectiveness in absorbing outgoing thermal infrared radiation.

we have imperfect knowledge about how the capacity of oceans (and other sinks) to absorb CO_2 will change as GHG concentrations change with consequent impacts on mean temperatures. This is one of the major sources of uncertainty in the current state of climate science.

9.5.3 How will climate change over the coming century and beyond?

Atmospheric concentrations of GHG are one major driver of global climate change. But those concentrations are not the sole driver. Looking back to Figure 9.13, one can identify several other factors that mediate this relationship. Leaving aside variations in incoming solar radiation, it is clear that any change to the reflection of either incoming solar radiation **above** the Earth's surface (such as the nature and extent of cloud cover, and water vapour) or **at** the Earth's surface (surface albedo effects, such as the extensiveness of snow and ice, and the nature of vegetative cover) will alter surface temperatures, for any given level of GHG concentration. Recent improvements in climate science have led to a better understanding of the roles played by water vapour, sea-ice dynamics and ocean heat transport, the latter particularly affecting the spatial variations in warming.[32] However, large uncertainties persist, particularly with regard to the feedback effects of changing cloud cover and their interaction with incoming radiation and aerosols.

9.5.3.1 Global mean temperature changes

We now turn to projections of global mean temperature changes implied by the various emissions scenarios. These projections are illustrated in Figure 9.15. The lines in the diagram are multi-model global averages of surface warming, relative to 1980–1999, for the scenarios A2, A1B and B1, shown as continuations of the 20th century simulations. Shading around those lines denotes the ±1 standard deviation range of individual model annual averages. The lowest line (after 2000) is for the experiment where concentrations were held constant at year 2000 values. The grey vertical bars at right indicate for 2100 the best estimate (solid line within each bar) and the likely range assessed for the six SRES marker scenarios.

Several points emerge from this picture. Leaving aside the experiment in which the GHG concentration was held constant at its 2000 level, all scenarios predict temperature increases by the year 2100 which dwarf those experienced in the previous century (about 0.5°C) or since pre-industrial times (about 1.0°C). Second, based on the likely ranges shown in the bars on the right-hand side, the minimum projected increase over the century to 2100 is 1°C and the maximum is over 6°C. Third, for several of the scenarios illustrated here, temperatures are far from having been stabilised by 2100, so

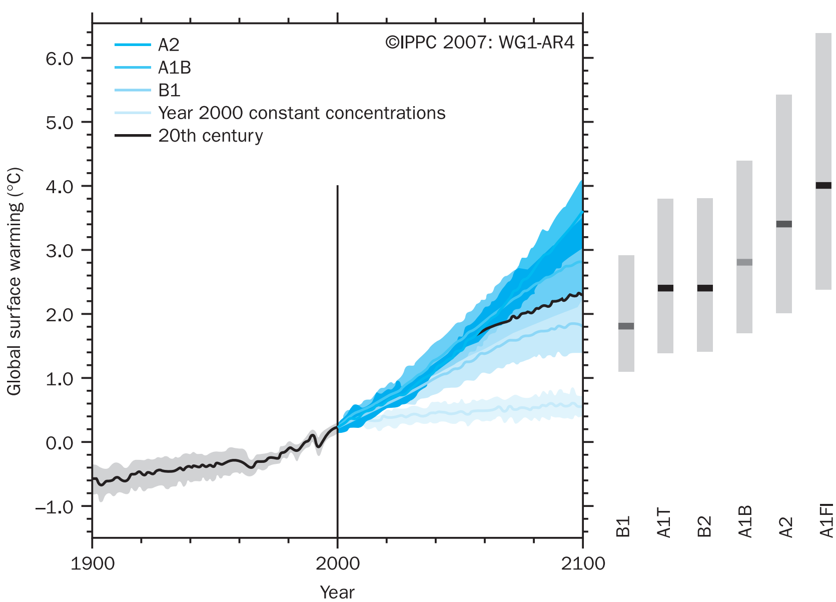


Figure 9.15 Multi-model averages and assessed ranges for surface warming. The reader should refer to the original image (available in full colour in the PowerPoint file and in the *Additional Materials* for Chapter 9) to ascertain the exact boundaries of the shaded areas.
Source: Figures 10.4 and 10.29, IPCC WG1, Summary for Policy Makers. Available online at http://www.ipcc.ch/pdf/assessment-report/ar4/wg1/ar4-wg1-spm.pdf

[32] For example, the Stern Review (2006) reports evidence that an increase in global mean temperature of above 1°C relative to pre-industrial levels is expected to weaken natural carbon storage and increase natural methane releases. An increase above 1.5°C is expected to begin irreversible melting of the Greenland ice sheet, and that increases beyond 3°C would increase risk of onset of abrupt, large scale shifts in the Earth's climate system, such as collapse of the Atlantic thermohaline circulation (THC, part of which is called the Gulf Stream or Atlantic conveyor) and the West Antarctic ice sheet.

non-intervention behaviour might entail temperature rises well above 6°C.

Plausible differences in assumptions about future emissions paths, therefore, generate considerable uncertainty about future global temperatures under business-as-usual behaviour. This uncertainty is (at least partially) independent of any uncertainty in the underlying physical science that has been referred to earlier. It also seems reasonable to conjecture that the prospect of a further mean temperature rise of 6°C or above, even where the probability of such an outcome is unknown, implies that taking no action to mitigate human-induced climate change can have no justification whatsoever.

9.5.3.2 Regional variations in climate

Our interest lies not only in global mean temperature changes, but also in their variation across space. This is typically done through the use of global circulation models (also called carbon cycle models), which simulate atmospheric and oceanic dynamic processes. Section 9.5.1 gave a brief account of the processes that generate these circulation processes. Simulations from such models generate information about what the mean temperature and its spatial variation would be at various points in time if GHG concentrations were to evolve along particular paths.

Unfortunately, a major limitation of current scientific understanding of the greenhouse effect relates to spatial disaggregation. It is generally agreed that regional variations in the degree of additional warming will show a large spread around the global average. And whilst the confidence with which regional predictions can be made has sharply increased in the past ten years, we still know relatively little about climate change at regional and national levels. But it is exactly this kind of information that is required in order to make well-founded estimates about the impacts of climate change. Some preliminary estimates about regional variations in temperature (and other indicators of climate) are given in the next section where we consider the impacts of climate change.

9.5.4 The impacts of climate change and the monetary value of potential damages

Two further matters need to be discussed before we are in a position to discuss policy choices: the potential impacts of global climate change, and the costs of abating GHG emissions. This section considers likely damage impacts over the century to 2100.

Table 9.6 lists a variety of potential impacts identified in the AR4 Reports and in the 2007 Stern Review. These impacts refer to changes expected under a range of emissions scenarios in which no *additional* mitigation measures are taken over the relevant time horizons. The impacts reflect not only projected changes in global mean temperatures, but also changes in precipitation and other climate characteristics, and consequent sea-level changes.

As can be seen from the table, there are very many pathways through which climate change can lead to possible future physical, biological, ecological and economic impacts. Some of these impacts are likely to be beneficial for humans, but others are expected to be adverse, in many cases seriously adverse, and in some cases potentially catastrophically damaging. For many potential impacts, though – particularly the ecological impacts – uncertainty about the magnitude of the impact remains pronounced, particularly the impacts at higher levels of temperature increase and at longer time horizons.

IPCC 2007 projections are for the global average sea-level to rise by between 0.18 and 0.59 metres over the century to 2100.[33] This alone could have catastrophic consequences for some parts of the world. Other alarming findings include the possibility that with global average temperature rising, several inner continental areas would experience higher probabilities of severe drought and soil degradation.

Natural systems can be very sensitive to climate change because of limited adaptive capacity. Many of these impacts are irreversible. Some of the systems at risk are identified in Table 9.6. It is likely that biodiversity decline will be intensified, at a rate which will depend on extent and rate of climate

[33] These figures are derived from simulations over the full range of SRES emissions scenarios, but exclude effects of rapid changes in ice flows.

Table 9.6 Potential future impacts of climate change

Potential impacts	Comments
Freshwater resources and their management	
By 2050, annual average river runoff and water availability are projected to increase by 10–40% at high latitudes and in some wet tropical areas, and decrease by 10–30% over some dry regions at mid-latitudes and in the dry tropics, some of which are presently water-stressed areas.	In some places and in particular seasons, changes will differ from these annual figures. Drought-affected areas will likely increase in extent. Heavy precipitation events, which are very likely to increase in frequency, will augment flood risk.
Water supplies stored in glaciers and snow cover are projected to decline, reducing water availability in regions supplied by meltwater from major mountain ranges	More than one-sixth of the world population currently lives in such areas. Adaptation procedures and risk management practices for the water sector are already being developed in some countries and regions.
Ecosystems	
The resilience of many ecosystems is likely to be exceeded this century by an unprecedented combination of climate change, associated disturbances and other global change drivers.	Examples of associated disturbances: flooding, drought, wildfire, insects, ocean acidification. Examples of other global change drivers: land-use change, pollution, over-exploitation of resources.
Approximately 20–30% of plant and animal species assessed so far are likely to be at increased risk of extinction if increases in global average temperature exceed 1.5–2.5°C. (Stern reports upper level of range as 50%.)	Systems at risk include glaciers, coral reefs, mangroves, boreal and tropical forests, polar and alpine ecosystems, prairie wetlands and remnant native grasslands.
Over the century, net carbon uptake by terrestrial ecosystems is likely to peak before mid-century and then weaken or even reverse, thus amplifying climate change.	
For increases in global average temperature exceeding 1.5–2.5°C and in concomitant atmospheric carbon dioxide concentrations, there are projected to be major changes in ecosystem structure and function, species' ecological interactions, and species' geographical ranges, with predominantly negative consequences for biodiversity, and ecosystem goods and services.	Ecosystem goods and services likely to be affected in this way include water and food supply. Beyond increase of 2.5°C, possible onset of collapse of part or all of Amazonian rainforest.
The progressive acidification of oceans due to increasing atmospheric carbon dioxide is expected to have negative impacts on marine shell-forming organisms and their dependent species.	Example: corals
Food, fibre and forest products	
Crop productivity is projected to increase slightly at mid- to high latitudes for local mean temperature increases of up to 1–3°C depending on the crop, and then decrease beyond that in some regions.	Beyond increase of 2.5°C rising number of people at risk from hunger. Beyond 4.5°C entire regions (such as one third of all Africa) expected to experience major declines in crop yields (Stern).
At lower latitudes, especially seasonally dry and tropical regions, crop productivity is projected to decrease for even small local temperature increases (1–2°C), which would increase the risk of hunger.	Increases in the frequency of droughts and floods are projected to affect local crop production negatively, especially in subsistence sectors at low latitudes. Adaptations such as altered cultivars and planting times allow low- and mid- to high-latitude cereal yields to be maintained at baseline for modest warming.
Globally, commercial timber productivity rises modestly with climate change in the short to medium term.	However, there will be large regional variability around the global trend.
Regional changes in the distribution and production of particular fish species are expected due to continued warming, with adverse effects projected for aquaculture and fisheries.	

Table 9.6 (*continued*)

Potential impacts	Comments
Coastal systems and low-lying areas	
Coasts are projected to be exposed to increasing risks, including coastal erosion, due to climate change and sea-level rise.	The effect will be exacerbated by increasing human-induced pressures on coastal areas.
Corals are vulnerable to thermal stress and have low adaptive capacity.	Increases in sea surface temperature of about 1–3°C are projected to result in more frequent coral bleaching events and widespread mortality, unless there is thermal adaptation or acclimatisation by corals.
Coastal wetlands including salt marshes and mangroves are projected to be negatively affected by sea-level rise.	This is especially likely where they are constrained on their landward side, or starved of sediment.
Many millions more people are projected to be flooded every year due to sea-level rise by the 2080s.	The numbers affected will be largest in the mega-deltas of Asia and Africa while small islands are especially vulnerable.
Beyond increase of 4.5°C sea-level rise threatens major world cities including Hong Kong, London, New York, Shanghai and Tokyo.	Adaptation for coasts will be more challenging in developing countries than in developed countries, due to constraints on adaptive capacity.
Densely populated and low-lying areas where adaptive capacity is relatively low, and which already face other challenges such as tropical storms or local coastal subsidence, are especially at risk.	
Industry, settlement and society	
The most vulnerable industries, settlements and societies are generally those in coastal and river flood plains, those whose economies are closely linked with climate-sensitive resources, and those in areas prone to extreme weather events, especially where rapid urbanisation is occurring. Where extreme weather events become more intense and/or more frequent, the economic and social costs of those events will increase, and these increases will be substantial in the areas most directly affected.	Costs and benefits of climate change for industry, settlement and society will vary widely by location and scale. In the aggregate, however, net effects will tend to be more negative the larger the change in climate. Poor communities can be especially vulnerable, in particular those concentrated in high-risk areas. They tend to have more limited adaptive capacities, and are more dependent on climate-sensitive resources such as local water and food supplies.
Health	
Projected climate change-related exposures are likely to affect the health status of millions of people, particularly those with low adaptive capacity, through: ▪ increases in malnutrition and consequent disorders, with implications for child growth and development; ▪ increased deaths, disease and injury due to heat waves, floods, storms, fires and droughts; ▪ the increased burden of diarrhoeal disease; ▪ the increased frequency of cardio-respiratory diseases due to higher concentrations of ground-level ozone related to climate change; and, ▪ the altered spatial distribution of some infectious disease vectors.	Climate change is expected to have some mixed effects, such as a decrease or increase in the range and transmission potential of malaria in Africa. Studies in temperate areas have shown that climate change is projected to bring some benefits, such as fewer deaths from cold exposure. Overall it is expected that these benefits will be outweighed by the negative health effects of rising temperatures worldwide, especially in developing countries. The balance of positive and negative health impacts will vary from one location to another, and will alter over time as temperatures continue to rise. Critically important will be factors that directly shape the health of populations such as education, health care, public health initiatives and infrastructure and economic development.

Source: Compiled from IPCC AR4 WG(2), 2007 and the Stern Review (2007)
Notes: Except where otherwise stated, changes refer to those projected over the century to 2100. All temperature changes are relative to the period 1980–1999. To express the change relative to the period 1850–1899, add 0.5°C.
It should also be borne in mind that the comments in this table that are attributed to Stern are not accepted by all climate change scientists.

change. The table also identifies several of the more sensitive human systems. Potential impacts which have been intensively studied include changes in crop yields (several studies expect adverse overall impacts on yields, but this result is far from certain), water availability (expected to worsen where water is already scarce), risks to human health and disease, and exposure to extreme events such as flooding.

There would be further induced impacts on the economic system, particularly in the energy and industry sectors, and in financial services.

Of great concern are impacts which may arise from threshold effects in climate change. These may include discontinuous changes in ocean circulation patterns, large reduction in Greenland and West Antarctic ice sheets, and accelerated global warming due to feedback effects such as release of carbon dioxide and methane from permafrost.[34] There remains much uncertainty about the timing and scale of such events, but their consequences would be catastrophic and largely irreversible.

A recurrent theme in climate-change impact studies is that those who are already relatively disadvantaged are likely to suffer the largest adverse impacts.[35] This arises partly because adaptation to climate change is a necessary part of any climate change policy – it is simply not feasible to prevent climate change from occurring. However, communities with low income and wealth will have the lowest ability to adapt. It also follows from the spatial distribution of climate change impacts. For example, the most serious impacts on farming are projected to occur in areas already experiencing high population growth and/or decreasing soil fertility. Table 9.7 gives examples of projected impacts for various regions of the world if no additional climate change control measures were to be adopted.

9.5.4.1 Damage valuation

Suppose that policy makers wished to identify an efficient (or what some writers call an optimal) level of GHG control. Conventional economic approaches would require that values in some common metric be put on the costs and benefits of climate change, or equivalently on the costs and benefits of control measures to reduce climate change. Either way, this would require some economic measure of the damages of climate change. Such a measure is usually expressed in monetary units.[36]

There are many ways in which this could be done. We shall explore these methods at some length in Chapters 12 and 13. The general technique is straightforward in principle. Once impacts have been estimated in their original 'physical' units, one assigns monetary values to those impacts using some technique which tries to proxy for what individuals are willing to pay to secure a benefit or willing to accept in compensation to avoid an impact that would otherwise occur.

It is generally accepted that this can be a legitimate exercise when done for small changes that take place within a single national economy. Even then, the caveat 'can be' is important, as we shall see in our discussions about valuation in Chapters 12 and 13. But it is an altogether more problematic exercise when attempted for large-scale changes, and/or for changes that occur in several countries simultaneously.

There are two problems here. The first is that conventional *operational* welfare measures are constructed on the assumption that changes are small, and do not move an economy far from its original resource allocation. Non-marginal changes can in principle be valued, but that is likely to be more difficult in practice than valuing marginal impacts. But marginal valuations may be inappropriate for large changes. The second problem involves a wider aspect of economic methodology, concerning how international comparisons can be made legitimately when the global distribution of income and wealth is far from what could be regarded as optimal.[37]

[34] For details about the release of carbon dioxide and methane as permafrost melts, see http://www.arctic.noaa.gov/essay_romanovsky.html

[35] A reflection of the huge uncertainty surrounding projections of the impacts of climate change is that even the claim that those who are already relatively disadvantaged are likely to suffer the largest adverse impacts is contentious. For example, poor people in cold countries might gain considerably.

[36] This requirement is needed even for much weaker criteria than an efficient policy. For example, even if one were merely to require that the benefits of control exceed the costs of control, damage evaluations would still be necessary.

[37] Similar considerations apply to the choice of discount rate. When a sequence of monetary flows is being summed over time to find the net present value of that series, the choice of discount rate can have a very large bearing – and sometimes dominating effect – on the outcome. It is noteworthy that the 2007 Stern Review selected its discount rate at least in part based on distributional and ethical considerations. In so doing it arrived at, and used, a discount rate much lower than those used by most other climate modellers. Stern was widely criticised by some economists for doing so, but has put up a strong defence for his choice. (See, for example, Stern's February 2008 letter to the Garnaut Climate Change Review ('Letter from Professor Lord Nicholas Stern', on Companion Website).) Readers should also note that there was a discussion of Stern's discounting in Chapter 3 of this textbook. See especially Box 3.1. See also Box 11.2.

Table 9.7 Examples of projected regional climate change impacts

Africa	▪ By 2020, between 75 and 250 million of people exposed to increased water stress due to climate change. ▪ By 2020, in some countries, yields from rain-fed agriculture reduced by up to 50%. Agricultural production and access to food in many African countries severely compromised. This further adversely affects food security and exacerbates malnutrition. Particularly at risk is the Sahel region of Africa, where extensive crop failures likely even for temperatures increases of less than 1°C. ▪ Towards the end of the 21st century, projected sea level rise will affect low-lying coastal areas with large populations. The cost of adaptation could amount to at least 5 to 10% of Gross Domestic Product (GDP). ▪ By 2080, an increase of 5 to 8% of arid and semi-arid land in Africa is projected under a range of climate scenarios.
Asia	▪ By the 2050s, decrease in freshwater availability in Central, South, East and South-East Asia, particularly in large river basins. ▪ Coastal areas, especially heavily populated mega delta regions in South, East and South-East Asia, will be at greatest risk due to increased flooding from the sea and, in some mega deltas, flooding from the rivers. ▪ Endemic morbidity and mortality due to diarrhoeal disease primarily associated with floods and droughts are expected to rise in East, South and South-East Asia due to projected changes in the hydrological cycle.
Australia and New Zealand	▪ By 2020, significant loss of biodiversity in some ecologically rich sites, including the Great Barrier Reef and Queensland Wet Tropics. ▪ By 2030, water security problems intensify in southern and eastern Australia and in New Zealand. ▪ By 2030, production from agriculture and forestry to decline over much of southern and eastern Australia, and over parts of eastern New Zealand, due to increased drought and fire. However, in New Zealand, initial benefits in some other regions. ▪ By 2050, ongoing coastal development and population growth in some areas of Australia and New Zealand exacerbate risks from sea-level rise and increases in the severity and frequency of storms and coastal flooding.
Europe	▪ Climate change is expected to magnify regional differences in Europe's natural resources and assets. Negative impacts will include increased risk of inland flash floods and more frequent coastal flooding and increased erosion (due to storminess and sea level rise). However, there are gains to some northern households arising from less severe coldness of temperature. Some areas of Europe are likely to have gains to tourism. ▪ Mountainous areas will face glacier retreat (or entire loss for smaller mountain glaciers), reduced snow cover and winter tourism, and extensive species losses (in some areas up to 60% under high emissions scenarios by 2080). ▪ In southern Europe, especially Mediterranean areas, climate change is projected to worsen conditions (high temperatures and drought) in a region already vulnerable to climate variability, and to reduce water availability, hydropower potential, summer tourism and, in general, crop productivity. ▪ Climate change is also projected to increase the health risks due to heat waves and the frequency of wildfires.
Latin America	▪ By mid-century, increases in temperature and associated decreases in soil water lead to gradual replacement of tropical forest by savanna in eastern Amazonia. Semi-arid vegetation will tend to be replaced by arid-land vegetation. ▪ Risk of significant biodiversity loss through species extinction in tropical Latin America. ▪ Productivity of some important crops to decrease and livestock productivity to decline, with adverse consequences for food security. In temperate zones, soybean yields are projected to increase. Overall, the number of people at risk of hunger to increase. ▪ Changes in precipitation patterns and the disappearance of glaciers are projected to significantly affect water availability for human consumption, agriculture and energy generation.
North America	▪ Warming in western mountains is projected to cause decreased snowpack, more winter flooding and reduced summer flows, exacerbating competition for over-allocated water resources. ▪ In the early decades of the century, moderate climate change is projected to increase aggregate yields of rain-fed agriculture by 5 to 20%, but with important variability among regions. Major challenges are projected for crops that are near the warm end of their suitable range or which depend on highly utilised water resources. ▪ Cities that currently experience heat waves are expected to be further challenged by an increased number, intensity and duration of heat waves during the course of the century, with potential for adverse health impacts. ▪ Coastal communities and habitats will be increasingly stressed by climate change impacts interacting with development and pollution.

Source: IPCC, 2007, AR4, WGII, 'Impacts, adaptation and vulnerability'. Available online at http://www.ipcc.ch/publications_and_data/publications_ipcc_fourth_assessment_report_wg2_report_impacts_adaptation_and_vulnerability.htm. Supplemented by additional information from Stern (2007)

We shall explain these problems at some length in discussing cost–benefit analysis in Chapter 11. It is noteworthy that the Report of Working Group III of the IPCC Second Assessment Group (IPCC, 1995c) did come up with numerical damage estimates. These estimates, though, generated considerable controversy (not all of it justified). Together with the ubiquity of climate itself, this may explain why the Third Assessment Group Report (IPCC(3), 2001) avoided giving any additional monetary evaluations of damages, and why such figures are also absent from AR4 Reports.

If these methodological problems have substance then they will have significant implications for international policy towards climate change. Impacts may well be very large, will bear down unequally between countries, and impact on a world in which the wealth distribution is extremely skewed. The size of possible impacts can be gauged from the fact that, for doubling of CO_2 concentration scenarios, studies typically place damages in the range 1% to 1.5% of GDP per year for developed countries, and 2% to 9% for developing countries. However, there is no reason to believe that concentrations will rise only to that extent, and as we have seen it is quite possible that average temperatures will eventually increase by more than 6°C. Damages would then be considerably greater than those just indicated.

Overall, as far as the regional distribution of damages arising from climate change is concerned, a commonly held view is that there will be few, if any, big winners but there will almost certainly be some very large losers. (See for example Nordhaus, 1990a, 2008; Hansen, 1990; Nordhaus and Boyer, 1999; and Stern, 2007.) Subscribers to that view often contend that that on average, damage is expected to be inversely related to per capita income. But each of these conjectures is contentious. It would be more accurate to say that the incidence of the impacts at an individual country level will depend on whether climate change propels a country closer to or further from its climate optimum. It is a moot point as to whether an assertion that there will be few big winners but almost certainly some very large losers or an assertion that damage is on average inversely related to per capita income is consistent with that more general principle.

What is less contentious is that those economies likely to suffer most tend to have the poorest resource base to implement policies that adapt to climate change and minimise the most serious forms of damage.

9.5.5 The costs of greenhouse gas reductions

The cost of achieving greenhouse gas reductions (either in terms of emission rates or in terms of atmospheric concentrations) will vary depending on the magnitude of the reduction being sought, how quickly the reduction is sought, and what method, or mix of methods, is being used to attain the reduction. Before we present some estimates of minimised GHG abatement costs, we first consider the options available for attaining GHG atmospheric concentration targets.

9.5.5.1 Options available for mitigating GHG atmospheric concentrations[38]

There are two ways to move towards a goal of reducing the rate of growth of atmospheric greenhouse-gas concentrations:

1. increase the capacity of sinks that sequester carbon dioxide and other greenhouse gases from the atmosphere;
2. decrease emissions of greenhouse gases below business as usual (thereby reducing GHG inflows to the atmosphere).

For any relatively non-stringent target, stabilisation of GHG concentrations could be achieved by the second of these alone. For more stringent targets, particularly where baseline emissions are high, some combination of the two methods would be required. Let us briefly explore the first route: increasing the capacity of terrestrial carbon sinks. Forests, agricultural lands and other terrestrial ecosystems offer significant atmospheric carbon reduction potential,

[38] The word 'mitigation' is used by the IPCC as a general term for what we have in this book usually called 'abatement' or 'emissions reduction'. It does not refer to reducing the impacts of any given level of GHG concentration.

operating through several channels. First, increased net planting rates and standing volumes would increase the amount of biomass that accumulates through natural growth; this biomass consists in large part of sequestered carbon. Secondly, some changes in the species or varieties mix of crops and other biomass can enhance the amount of carbon that is stored. Thirdly, changes in agricultural practice and patterns of land use can conserve existing stocks of carbon more effectively, preventing its discharge into the atmosphere. For example, 'no till' agricultural practices can retain carbon content relative to conventional plough-then-sow methods.

IPCC estimates suggest that the potential is large but nevertheless limited. There are two kinds of limit. The first concerns the feasible size of the sinks. The 2001 IPCC Report saw a total cumulative sequestration potential of 100GtC by 2050, equivalent to between 10% and 20% of potential fossil-fuel emissions during that period. The most recent IPCC AR4 Report, however, points to substantially higher potential from a wider variety of carbon sinks than were previously thought to exist, with that potential rising sharply as the carbon price increases. The second limit concerns duration through time. A once-for-all forestation project would store carbon for the lifetime of the timber, but once that timber decays or is burnt, its carbon is returned to other sinks, including the atmosphere. Put another way, larger biomass stocks today imply greater potential flows to the atmosphere in the future. Without proper management, therefore, this approach may merely reschedule the temporal pattern of carbon flows to and from the atmosphere, and pose a risk of substantially greater CO_2 emissions in the longer-term future. A *permanent* reduction of atmospheric carbon stocks would require, loosely speaking, a permanent increase in the stock of the biomass in question.[39]

Even if this option is feasible, it may not be economically sensible. That depends on the costs of sequestering carbon in this way compared to the costs of reducing equivalent amounts of carbon emissions. Such cost comparisons need also to take into consideration opportunity costs of the land on which biomass is accumulated.[40] A sensible approach might be to look for sequestration projects that generate synergies, by being complementary to other activities or land uses, such as wildlife or biodiversity reserves, or recreational activities. The IPCC argues that the best long-term goal is to substitute wood for other materials in building and manufacturing to make carbon storage in timber more long lasting.

Preliminary IPCC estimates suggest that the (undiscounted) marginal costs of these schemes are in the region of \$0.1/tC to \$20/tC in several tropical countries; and from \$20/tC to \$100/tC in non-tropical countries. However by not including opportunity costs of land, infrastructure and some other associated 'indirect costs', these figures underestimate full long-run marginal costs. Moreover, marginal costs will rise as the best carbon sequestration projects are taken up.

As a practical proposition, it is now generally accepted that increased carbon sinks can only partially offset expected fossil-fuel emissions, although they do have a very important role to play. Given the limited scope for larger terrestrial sinks, a large component of any programme must involve GHG emissions reductions. How *large* these would have to be is discussed below. To think about *how* they might be achieved, it is worth looking at the following identity (which assumes for simplicity that carbon emissions arise entirely in energy use)[41]

$$M \equiv \frac{M}{N} \cdot \frac{N}{E} \cdot \frac{E}{N} \cdot \frac{N}{Y} \cdot \frac{Y}{N} \cdot N$$

[39] An alternative way of sequestering carbon stocks from the atmosphere and locking them up in a terrestrial sink is by means of carbon capture and storage (CCS). As conventionally used, this term applies to non-biological means of capturing carbon emissions at source and storing them. An IPCC Special Report (IPCC, 2005, section 8.3.3) estimated that the economic potential of CCS could be between 10% and 55% of the total carbon mitigation effort until year 2100. However, capturing and compressing CO_2 is energy intensive and would increase the fuel needs of a coal-fired plant by between 25% and 40%, substantially increasing generation costs. Whether this is a more cost-effective option than mitigation by other means remains to be seen, as CCS systems are not yet present on a commercial scale, and the impact on costs of anticipated technological improvements can only be guessed. Some further reading on CCS is suggested at the end of this chapter.

[40] The opportunity cost is the highest value alternative use of that land.

[41] The use being made here of an accounting identity is similar to our discussion in Chapter 2 of the IPAT identity with respect to carbon emissions.

where M is total carbon emissions, N is total population, E is total energy use, and Y is total output (or national income). The terms on the right-hand side can be rearranged so that the identity reads as

$$M \equiv \frac{M/N}{E/N} \cdot \frac{E/N}{Y/N} \cdot \frac{Y}{N} \cdot N$$

or using *per capita* notation

$$M \equiv \frac{m}{e} \cdot \frac{e}{y} \cdot y \cdot N$$

where $m = M/N$ is emissions per person, $e = E/N$ is the energy used per person, and $y = Y/N$ is output per person. Consider the last of these three forms of the identity. What does this tell us? Total emissions will fall if any term on the right-hand side falls, other things being equal. Hence we can arrive at the following conclusions (assuming that other things stay constant in each case):

1. Emissions will fall if m/e – which reflects the emissions intensity of energy production – can be reduced. This could be achieved by changes in fuel mix (from fossil to renewable energy, for example), or by emissions reduction measures in energy generation (such as carbon-capture technology) being put in place.
2. Emissions will fall if e/y – the energy intensity of output – falls. This could be achieved by changes in output composition, or by producing output in more energy-conserving ways.
3. Emissions will fall if y, *per capita* national income, falls.
4. Emissions will fall if N falls. At the global level, there is no prospect of this for several decades. Indeed, for some time rising global population will be a contributory factor to growing emissions. But as Chapter 2 showed, global population might fall by the end of the twenty-first century, and certainly cannot grow without limit indefinitely.

Items 1 and 2 are the most suggestive of ways in which emissions might be reduced, and are those on which most discussions focus. The final two items are of less policy relevance, partly because it seems unlikely that any economy would be willing to accept a cut in per capita income if other alternative ways of reducing emissions existed that did not entail real income cuts. Using population size as a means of controlling harmful emissions likewise seems unreasonable to say the least. But, multiplying per capita income by population size gives the total value of national income. This serves to remind us of the importance of scale *per se* when thinking about our impacts on the environment: other things being equal, the larger the scale of human economic activity the greater will be the adverse impacts on the natural environment. It would be a useful exercise to consider how a carbon tax (or a system of tradable emissions permits) might alter the components we have been discussing. We leave this as an exercise for the reader.

One must be wary of trying to extract too much from accounting identities of this kind. The identity *per se* is true by construction. But the use made of that identity requires careful thought. There are at least two reasons for this. First, each item in the identity is an aggregate of what may in practice be heterogeneous components. And second, one must allow for the possibility that items in the different aggregate components may not be independent of one another, and so invoking the 'other things remaining equal' assumption might not be valid. For example, there are many ways in which energy efficiency could be improved at relatively low cost, thereby decreasing the energy intensity of output, e/y. But greater energy efficiency will tend to lower the relative price of energy. If energy demand elasticities were sufficiently high, substitution effects might increase the demand for energy, possibly changing the value of y and making the sign of the change in e/y ambiguous. None of this means that the identity is of no use in providing insight into how policy to reduce GHG emissions might operate. It just means that we need to be careful in how we draw inferences from it.

9.5.5.2 Some general results about the costs of attaining GHG emissions or atmospheric concentration targets

In the period since 1990, there has been something close to an industry within the economics profession researching the likely costs of greenhouse gas abatement. Rather than attempting to survey this literature, we shall report some findings of the IPCC third working group dealing with 'Mitigation of Climate Change', which provides a fairly comprehensive statement of what is currently known.[42] Some key results emerge from that document, with important policy implications.

1. The cost of achieving any given target in terms of levels of allowable GHG emissions or stabilised GHG concentrations increases as the magnitude of the emissions or concentration target declines.
2. Other things being equal, the cost of achieving any given target increases the higher are baseline (i.e. uncontrolled) emissions over the time period in question.
3. The cost of achieving any given target varies with the date at which targets are to be met, but does so in quite complex ways. It is not possible to say in general whether fast or early control measures are more cost-effective than slow or late controls.
4. There is some scope for GHG emissions to be reduced at zero or negative net social cost. The magnitude of this is uncertain. It depends primarily on the size of three kinds of opportunities and the extent to which the barriers limiting their exploitation can be overcome:

   - overcoming market imperfections (and so reducing avoidable inefficiencies);
   - ancillary or joint benefits of GHG abatement (such as reductions in traffic congestion);
   - double dividend effects (see Chapter 5).

   To give some indication about magnitudes, IPCC WG III (2007c) estimates that abatement opportunities of the kind given in the first two bullet points listed above have the potential to reduce emissions by around 6 $GtCO_2$-eq/yr in 2030, approximately 14% of such emissions in 2000.
5. Abatement costs will be lower the more cost-efficiently that abatement is obtained. This implies several things:

   - Costs will be lower for strategies that focus on all GHGs, rather than just CO_2, and are able to find cost-minimising abatement mixes among the set of GHGs. Put another way, it is not just carbon emissions or concentrations that matter.
   - Costs will be lower for strategies that focus on all sectors, rather than just one sector or a small number of sectors. Thus, for example, while reducing emissions in energy production is of great importance, the equi-marginal principle suggests that cost minimisation would require a balanced multi-sectoral approach.
   - The more 'complete' is the abatement effort in terms of countries involved, the lower will be overall control costs. This is just another implication of the equi-marginal cost principle, and it also is necessary to minimise problems of carbon (or other GHG) leakage. This 'leakage' issue has been at the heart of concerns about the limited effectiveness of emissions constraints under the first phase of the Kyoto Protocol. Under its provisions, mandatory controls relate only to the so-called Annex 1 countries. Quite apart from the fact that the non-Annex 1 countries are under no obligations to abate GHG emissions the net emissions reductions achieved will be lower than the gross reductions achieved by Annex 1 countries to the extent that 'carbon leakage' occurs. For example, tighter constraints on, or increasing costs of, carbon emissions in some locations may result in geographical relocations of some industries and changes in patterns of trade. Estimates of the extent of carbon leakage vary widely, but are thought to be in the order of 5% to 20% of Annex 1 constraints. More generally, free-rider considerations mean that without complete coverage, something close to an open access outcome may emerge, with the self-imposed limitations of some parties being offset by increased use of the atmospheric 'common' by others. Chapter 10 provides a more extensive analysis of these and related matters.

[42] This report, IPCC WG III (2007), is available in full online at http://www.ipcc.ch/publications_and_data/publications_ipcc_fourth_assessment_report_wg3_report_mitigation_of_climate_change.htm

- The above comments imply that in principle achieving targets at least cost could be brought about by the use of a set of uniform global GHG taxes. A uniform carbon tax would bring about carbon reductions cost-effectively, but would not generate a least-cost multi-gas abatement outcome. Alternatively, use could be made of a set of freely tradable net emissions licences (one set for each gas, with tradability *between* sets at appropriate conversion rates), with quantities of licences being fixed at the desired cost-minimising target levels.

6. Climate-change decision making is essentially a sequential process under uncertainty. The value of new information is likely to be very high, and so there are important quasi-option values that should be considered. We shall explain in Chapter 13 what is meant by a quasi-option value.

9.5.5.3 Numerical estimates of mitigation potential and mitigation costs[43]

In the next two sections, we summarise some of the quantitative results concerning climate change mitigation potential and costs given in the IPCC WG III Report (2007c). Much of the IPCC discussion about mitigation potential and costs is framed in terms of 'mitigation scenarios'. These are not to be confused with 'emissions scenarios'. The latter is designed for the purpose of projecting emissions *in the absence of any additional GHG control measures*. Emissions scenarios, and their associated emissions paths, provide baseline, reference, or 'business-as-usual' benchmarks. In contrast a *mitigation scenario* is used to project net changes in emissions or concentrations of GHGs under the assumption that *a particular set of GHG control measures are adopted*.

IPCC reports have chosen to present their findings about GHG mitigation potential and minimised mitigation costs for two time horizons: what it terms 'short and medium term mitigation' (up to 2030) and 'long term mitigation' (beyond 2030). Before proceeding, it will be worth reviewing IPCC 2007 baseline 'no additional control' emissions projections for the years 2030 and 2100. These are shown in Figure 9.16. The figure provides the emissions from the six illustrative SRES scenarios. It also provides the frequency distribution of the emissions in the post-SRES scenarios (5th, 25th, median, 75th, 95th percentile). F-gases cover HFCs, PFCs and SF6.

9.5.5.4 Short to medium term GHG mitigation: estimated mitigation costs for the period to 2030

Table 9.8 gives projections of mitigation costs for the period up to 2030. IPCC draws a distinction between 'top-down' and 'bottom-up' research studies. It defines these terms in the following way:

> *Bottom-up* studies are based on assessment of mitigation options, emphasizing specific technologies and regulations. They are typically sectoral studies taking the macro-economy as unchanged. *Top-down* studies assess the economy-wide potential of mitigation options. They use globally consistent frameworks and aggregated information about mitigation options and capture macro-economic and market feedbacks.

Many of the bottom-up studies incorporated in the IPCC findings suggest large opportunities for 'no regret' policies, that imply zero, or even negative, mitigation costs. It is this which explains why a zero carbon tax in the upper section of Table 9.8 results in substantial amounts of emissions abatement. These zero or negative mitigation cost possibilities are not found in top-down studies. The reason for this is that top-down studies assume, in effect, that

[43] IPCC uses the word 'mitigation' throughout its reports. The Glossary to IPCC WG III (2007) defines mitigation as: 'Technological change and substitution that reduce resource inputs and emissions per unit of output. Although several social, economic and technological policies would produce an emission reduction, with respect to climate change, mitigation means implementing policies to reduce GHG emissions and enhance sinks.' This textbook uses the word 'abatement' to refer actions that lead to an emissions reduction, and so is a subset of what IPCC means by mitigation. In this and subsequent sections we shall follow the IPCC convention of using the word mitigation rather than abatement when it refers to one or both of emissions reduction and sink enhancement. We shall use abatement only when the context is referring specifically to emissions reduction.

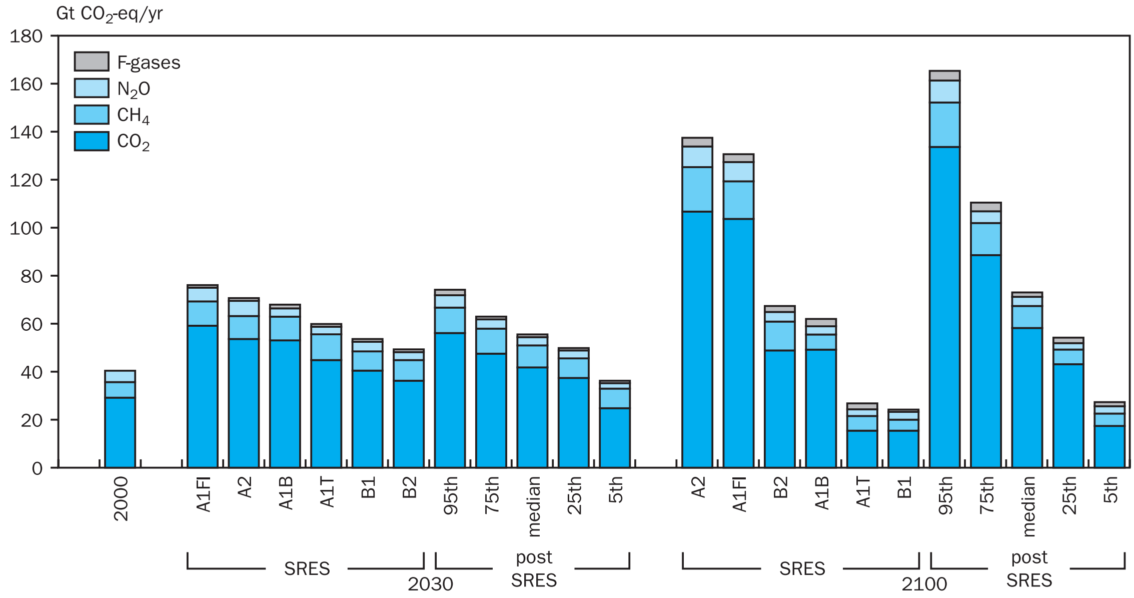


Figure 9.16 Global GHG emissions for 2000 and projected baseline emissions for 2030 and 2100 from IPCC SRES and the post-SRES literature

The figure provides the emissions from the six illustrative SRES scenarios. It also provides the frequency distribution of the emissions in the post-SRES scenarios (5th, 25th, median, 75th, 95th percentile), as covered in Chapter 3. F-gases cover HFCs, PFCs and SF6.

Source: IPCC WG III Report (2007c)

Table 9.8 Global economic mitigation potential in 2030 estimated from bottom-up and top-down studies

Table SPM.1: *Global economic mitigation potential in 2030 estimated from bottom-up studies.*

Carbon price (US$/t$CO_2$-eq)	Economic potential (GtCO_2-eq/yr)	Reduction relative to SRES A1 B (68 GtCO_2-eq/yr) (%)	Reduction relative to SRES B2 (49 GtCO_2-eq/yr) (%)
0	5–7	7–10	10–14
20	9–17	14–25	19–35
50	13–26	20–38	27–52
100	16–31	23–46	32–63

Table SPM.2: *Global economic mitigation potential in 2030 estimated from top-down studies.*

Carbon price (US$/t$CO_2$-eq)	Economic potential (GtCO_2-eq/yr)	Reduction relative to SRES A1 B (68 GtCO_2-eq/yr) (%)	Reduction relative to SRES B2 (49 GtCO_2-eq/yr) (%)
20	9–18	13–27	18–37
50	14–23	21–34	29–47
100	17–26	25–38	35–53

Source: IPCC WG III (2007c)

what we actually observe is cost minimising and x efficient.[44]

The table shows the estimated amounts of mitigation that would go with step changes in the price of carbon emissions. The carbon prices correspond to, and would be manifested by, either a uniform global carbon dioxide tax or to the equilibrium price of carbon dioxide emissions permits that would result from a global system of freely tradable emissions licence being fixed at the reduced emissions levels implied in the table.

IPCC defines 'mitigation potential' as the scale of GHG reductions that could be made, relative to emission baselines, for a given level of carbon price. It uses two measures of mitigation potential: 'market potential' is the mitigation potential based on private costs and private discount rates, which might be expected to occur under forecast market conditions, including policies and measures currently in place, noting that barriers limit actual uptake. 'Economic potential' is the mitigation potential, which takes into account social costs and benefits and social discount rates, assuming that market efficiency is improved by policies and measures and barriers are removed. Table 9.8 shows economic potential, while the IPCC reports also estimates market potential (not shown here).

Looking at these estimates, we see that top-down studies suggest that a globally applicable carbon tax of \$50/tCO_2-eq would yield an 'economic potential' GHG reduction in the range 14 to 23 gigatonnes of carbon dioxide equivalent greenhouse gases annually by the year 2030. This corresponds to a reduction in emission of between 35% and 53% of those projected for the year 2030 under the 'no additional intervention' baseline case of SRES Scenario B2.

Knowledge of the carbon tax and resulting mitigation levels allows one to estimate mitigation costs in the conventional format of the percentage of GDP lost per annum. We shall return to this matter shortly. Before we do so, some words of caution are warranted about the figures quoted for mitigation quantities. 'Economic potential' makes rather strong assumptions. In particular, it supposes that market efficiency is improved by the chosen policies and that barriers to policy implementation are removed. Estimates are also derived in the main from global least cost optimisation models that assume universal emissions trading, efficient markets, and no transaction cost. Hence they assume perfect implementation of mitigation measures. Mitigation costs (and so carbon taxes) will increase to the extent that inefficiencies exist and persist, barriers are not removed, transaction costs are positive, or implementation is not universal (i.e. some regions, sectors, options or gases are excluded). Equivalently, for any given carbon tax, reductions achieved would be lower. On the other hand, net mitigation costs (or implied carbon prices) will decrease with lower baseline emissions paths, carefully chosen recycling of revenues from carbon taxes and auctioned permits, and if induced technological learning is included. Furthermore, the IPCC-referenced models do not consider co-benefits of mitigation measures.

What are estimated mitigation costs as of 2030 in the conventional format of the percentage of GDP lost per annum? IPCC estimates that costs consistent with emissions trajectories towards stabilisation between 445 and 710 ppm CO_2-equivalent lie between a 3% decrease of global GDP and a small increase, compared to the baseline.[45] However, if revenues from carbon taxes or auctioned permits

[44] Such an 'assumption' may be made for several reasons. First, one may believe it to be true. Second (and more plausibly), it may be justified on the grounds of being a 'reasonable approximation'; this is still a very strong assumption, as it accords second-order ranking to any existing inefficiencies. Third, and most likely, it is done for reasons of simplification and tractability: the models used in top-down studies may not be amenable to solution without such a simplification. In all cases, making the full efficiency assumption is at best a moot point. And it may be far worse than that, assuming away something of real substance.

Many of the top-down studies make use of applied (or computable) general equilibrium models, also known as CGE models. We discussed such models earlier in Chapter 8.

[45] The ability to link GDP losses to GHG atmospheric concentration rates comes from the fact that the estimates in the lower half of Table 9.8 were derived from stabilisation scenarios (i.e. simulation runs towards long-run stabilisation of atmospheric GHG concentration).

are ring-fenced and used to promote low carbon technologies or reform of existing taxes, or if climate change policy induces enhanced technological change, the GDP costs may be substantially lower than the figures just quoted.[46]

Estimates are also available for the extent of emissions reductions achievable at 'reasonable cost' from individual economic sectors, and are summarised in Table 9.9.[47] These figures imply that total reductions in annual emissions of around 60% of 1990 levels are possible, a substantial proportion of which are claimed to be available at negative real cost. However, realising these gains makes very strong assumptions about future technological improvements and international technology transfer.

9.5.5.5 Long-term GHG mitigation, for stabilised GHG concentrations: estimated mitigation costs for the period after 2050

IPCC organises its findings about long-term climate change mitigation largely in terms of what is necessary for the stabilisation of atmospheric concentration of GHGs at particular levels. For each such level, the IPCC best estimate of climate sensitivity allows one to map that stabilisation target into its associated equilibrium global mean temperature increase. Six categories of GHG stabilisation target are described in Table 9.10. Each of those categories would necessitate a particular mix and timing of mitigation efforts. Category 1, with the lowest concentration target, requires the most stringent set of control measures, and requires that control measures are introduced at the earliest points in time. As one moves from category 1 to category 7, the stringency of required controls and their timing is progressively relaxed.[48]

Table 9.10 also shows IPCC 2007 estimates of the required emissions levels for each stabilisation scenario, and estimates of GDP losses (consisting of mitigation costs) and carbon prices that would be required to reach the stabilisation target in question.

To stabilise atmospheric concentration of GHGs, emissions would need to peak and decline thereafter. It is evident from Table 9.10 that stabilization at lower concentration and related equilibrium temperature levels advances the date when emissions need to peak, and requires greater emissions reductions by 2050. These aspects are shown more clearly in Figure 9.17, in which the likely range of projection uncertainty associated with each category is also provided. One should also bear in mind that the reported global mean temperature increases in Table 9.10 are reached well after concentrations are stabilised. For the majority of scenarios assessed, stabilisation of GHG concentrations occurs between 2100 and 2150.[49]

Table 9.10 and Figure 9.17 ignore the possibility of substantial feedbacks between the carbon cycle and climate change. There is now reason to believe that these feedbacks will increase the fraction of emissions that remains in the atmosphere as the climate system warms. As a result, the emission reductions to meet a particular stabilisation level of atmospheric carbon dioxide concentration reported in the mitigation studies assessed here might be underestimated, possibly substantially so.

For lower stabilisation levels, scenarios put more emphasis on the use of low-carbon energy sources, such as renewable energy and nuclear power, and the use of CO_2 capture and storage (CCS). In these scenarios improvements of carbon intensity of energy supply and the whole economy need to be much faster than in the past.

[46] Ring-fencing in this way is sometimes referred to as hypothecation and historically is not much liked by some treasuries (such as that in the UK).

[47] Given that these estimates are bottom-up, 'engineering' approaches, they may not be reliable when done at large scales.

[48] For reference purposes, the CO_2 concentration rate of 550 ppm has proved to be of particular interest. This falls within the Category IV range. It corresponds to an approximate doubling of CO_2 concentrations since pre-industrial times. Many research teams have chosen to study this objective.

[49] A large majority of studies that form the basis for these projections concentrate on carbon reduction strategies alone. Relatively few multi-gas studies exist. However, to allow comparison between CO_2-only and multi-gas mitigation, Table 9.10 and Figure 9.17 have been constructed so that each category corresponds to one particular range of CO_2 equivalent atmospheric concentrations, whether that would be achieved by CO_2 reduction alone or via a reduced mix of different gases.

Table 9.9 Key mitigation technologies and practices by sector

Sector	Key mitigation technologies and practices currently commercially available	Key mitigation technologies and practices projected to be commercialized by 2030	Economic potential mitigation at 2030, for carbon tax of up to US\$100/tCO_2-eq
Buildings	Efficient lighting; more efficient electrical appliances and heating and cooling devices; improved cook stoves, improved insulation; passive and active solar design for heating and cooling; alternative refrigeration fluids, recovery and recycle of fluorinated gases.	Integrated design of commercial buildings including technologies, such as intelligent meters that provide feedback and control; solar PV integrated in buildings.	5.3–6.7 Gt CO_2-eq/yr
Transport	More fuel efficient vehicles; hybrid vehicles; cleaner diesel vehicles; biofuels; modal shifts from road transport to rail and public transport; non-motorised transport; land-use and transport planning.	Second generation biofuels; higher efficiency aircraft; advanced electric and hybrid vehicles with more powerful and reliable batteries.	1.6–2.5 Gt CO_2-eq/yr
Energy supply	Improved supply and distribution efficiency; fuel switching from coal to gas; nuclear power; renewable heat and power (hydropower, solar, wind, geothermal and bioenergy); combined heat and power; early applications of Carbon Capture and Storage (CCS, e.g. storage of removed CO_2 from natural gas).	CCS for gas, biomass and coal-fired electricity generating facilities; advanced nuclear power; advanced renewable energy, including tidal and waves energy, concentrating solar, and solar PV.	2.4–4.7 Gt CO_2-eq/yr
Industry	More efficient end-use electrical equipment; heat and power recovery; material recycling and substitution; control of non-CO_2 gas emissions; and a wide array of process-specific technologies.	Advanced energy efficiency; CCS for cement, ammonia, and iron manufacture; inert electrodes for aluminium manufacture.	2.5–5.5 Gt CO_2-eq/yr
Agriculture	Improved crop and grazing land management to increase soil carbon storage; restoration of cultivated peaty soils and degraded lands; improved rice cultivation techniques and livestock and manure management to reduce CH_4 emissions; improved nitrogen fertilizer application techniques to reduce N_2O emissions; dedicated energy crops to replace fossil fuel use; improved energy efficiency.	Improvements of crops yields.	2.3–6.4 Gt CO_2-eq/yr
Forestry/Forests	Afforestation; reforestation; forest management; reduced deforestation; harvested wood product management; use of forestry products for bioenergy to replace fossil fuel use.	Tree species improvement to increase biomass productivity and carbon sequestration. Improved remote sensing technologies for analysis of vegetation/soil carbon sequestration potential and mapping land use change.	1.3–4.2 Gt CO_2-eq/yr
Waste management	Landfill methane recovery; waste incineration with energy recovery; composting of organic waste; controlled wastewater treatment; recycling and waste minimization.	Biocovers and biofilters to optimize CH_4 oxidation.	0.4–1.0 Gt CO_2-eq/yr

Source: IPCC (2007) WG III, Synthesis Report – Summary for Policy Makers. Table SPM.3 and Figure SPM.6

Table 9.10 Characteristics of some recent stabilisation scenarios

Category	CO_2 Concentration (b) (ppm)	CO_2-eq Concentration (b) (ppm)	Global mean temperature increase above preindustrial at equilibrium, using 'best estimate' climate sensitivity (a), (b) (°C)	Peaking year for CO_2 emissions	Change in global CO_2 emissions in 2050 (% of 2000 emissions)	GDP loss (%) in 2100 Central estimate (c)	GDP loss (%) in 2100 Range from studies reported by IPCC (d)	Carbon Tax in 2100 (US \$/t CO_2) (c)	Carbon Tax in 2100 (US \$/t CO_2) Range from studies reported by IPCC (d)
I	350–400	445–490	2.0–2.4	2000–2015	−85 to −50	5.4	na	300	na
II	400–440	490–535	2.4–2.8	2000–2020	−60 to −30	4.5	na	200	na
III	440–485	535–590	2.8–3.2	2010–2030	−30 to +5	3.6	−0.1 to 6.5	190	35 to 350
IV	485–570	590–710	3.2–4.0	2020–2060	+10 to +60	2.5	−1.6 to 5.0	112	25 to 200
V	570–660	710–855	4.0–4.9	2050–2080	+25 to +85	2.0	0.3 to 3.0	100	na
VI	660–790	855–1130	4.9–6.1	2060–2090	+90 to +140	1.6	na	80	na

Notes:

(a) The equilibrium climate sensitivity is a measure of the climate system response to sustained radiative forcing. It is not a projection but is defined as the global average surface warming following a doubling of carbon dioxide concentrations. The best estimate of climate sensitivity is 3°C.

(b) Note that global mean temperature at equilibrium is different from expected global mean temperature at the time of stabilisation of GHG concentrations due to the inertia of the climate system.

(c) The point estimates given here are the authors' own judgement of the central tendency of estimates, based on the set of data shown graphically in Figure 3.25 of WGIII Full Report. They are **NOT** values reported explicitly in the text of the IPCC Report. As such, they must be treated with caution, and are included only to indicate how costs are likely to vary as the stringency of the stabilisation target changes.

(d) The ranges reported here are 80th percentile ranges (lying between the 10th and 90th percentile), being those given on page 206 of WGIII Full Report. The term na refers to case where insufficient reports were available for that category to allow for comparison with those that are reported.

Source: Table SPM.5, AR4 WG111, SPM

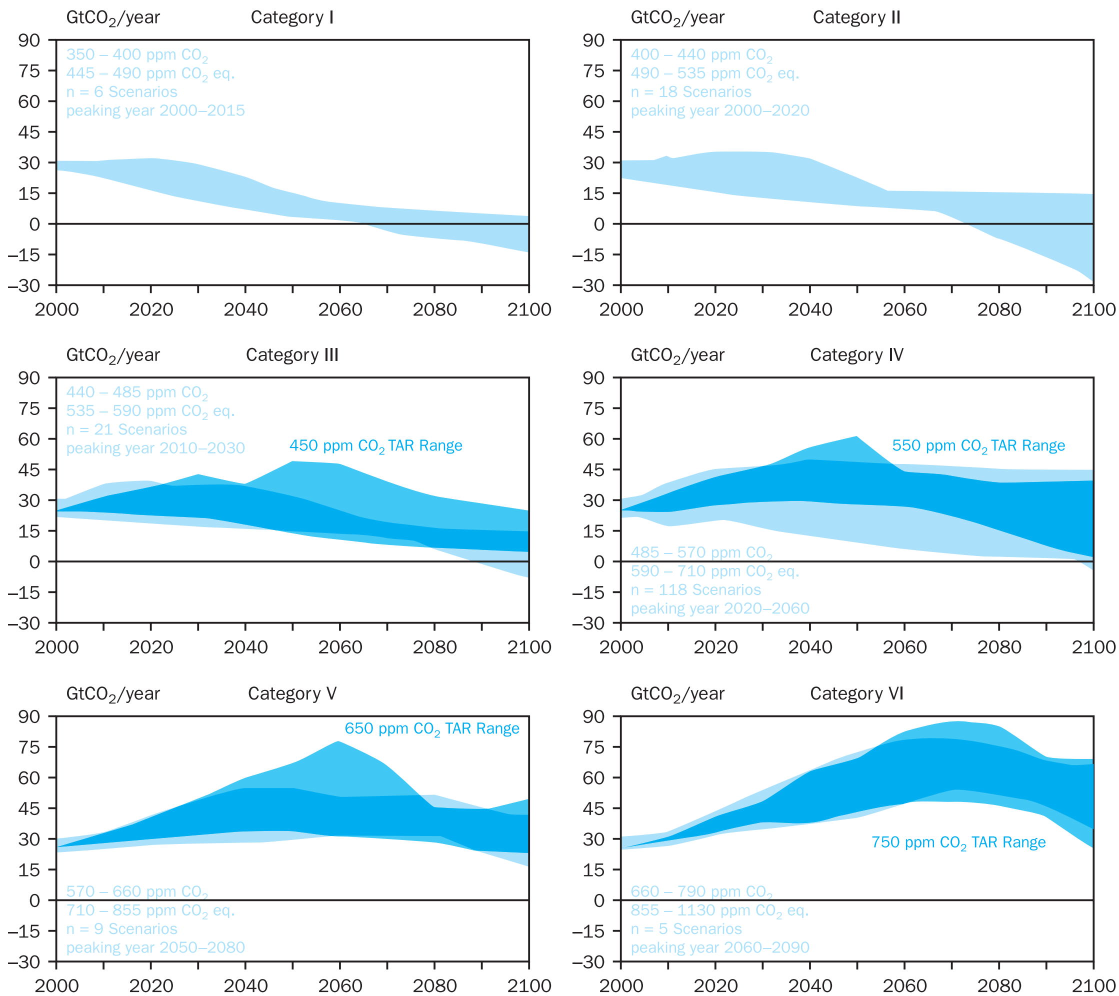


Figure 9.17 Emissions pathways of mitigation scenarios for alternative groups of stabilisation targets
Notes: The lightly shaded areas give the projected CO_2 emissions for the recent mitigation scenarios developed post-TAR. Mid-tone and heavily shaded areas depict the range of more than 80 TAR stabilization scenarios. Note that heavily shaded areas denote areas of overlap between lightly-shaded and mid-tone areas (Morita *et al.*, 2001).

Category I and II scenarios explore stabilization targets below the lowest target of the TAR. To reach the lower stabilization levels some scenarios deploy removal of CO_2 from the atmosphere (negative emissions) using technologies such as biomass energy production utilizing carbon capture and storage.
Source: Figure 3.17, WG III Full report, page 199. Based on Nakicenovic *et al.* (2006) and Hanaoka *et al.* (2006)

9.5.6 Policy Analysis

9.5.6.1 On what basis might choices be made?

1. Optimising as the basis for policy choices

A standard way in which economists derive policy recommendations is by means of an optimisation exercise. As we shall see later (in Chapters 14 to 18 on resource economics), to analyse stock pollution problems one begins by specifying an objective (or economic welfare) function, typically the present value of utility or consumption over some planning horizon. Relevant constraints are then identified and stated mathematically. One or more variables in the optimal growth model are instrumental (or policy) variables; it is these variables that the policy maker is assumed to have under its control. Policy choices emerge from the maximisation of economic welfare subject to the appropriate constraints. The outputs of the optimisation exercise yield:

1. a time path of quantities for the control or choice variable (or variables) that maximise the objective function subject to the operative constraints;
2. a time path of shadow prices, which are the 'dual' of the optimimising quantities referred to in 1. above. These shadow prices are usually interpretable as 'efficient' tax rates, such that in a decentralised market economy their imposition would lead agents to behave in a way that would generate the socially efficient optimal outcome.

There are many such intertemporal optimisation models of climate change policy. One of these is the Nordhaus DICE-2007 model. The structure of that model is described and explained briefly in Box 16.1 in Chapter 16 and at more length in the Word file *DICE-2007* in the *Additional Materials* for Chapter 10, and you may find it useful at this point to read those pages. In this section we take you through some of the DICE-2007 model policy recommendations.

In essence, the objective function in DICE-2007 is the present value of global consumption. Damages from GHG emissions reduce consumption possibilities, as do the costs of GHG abatement. The model allows the user to identify emissions abatement choices (the policy instruments) that maximise the present value of global consumption, net of GHG damages and abatement costs, over horizons of up to 200 years or so ahead. Nordhaus calls such a set of policy choices the 'optimal' policy.

DICE-2007 finds the net present value (NPV) benefit of the optimal policy, relative to a business as usual or 'uncontrolled' baseline, to be $3 trillion.[50] This and other characteristics of the optimal policy are shown in the second row of Table 9.11 (the row labelled 'Optimal'). Global temperature rises over its 1900 level by 2.6°C in 2100 and by 3.4°C in 2200. Despite the fact that substantial amounts of climate change mitigation are taking place, the present value of climate change damages are still very large, at $17.3 trillion. But those damages would be $22.6 trillion in the uncontrolled baseline case, and so are cut by $5.3 trillion. However, the PV of abatement costs are $2.2 trillion; when this figure is deducted from the $5.3 trillion fall in climate change damages, we arrive at the NPV of $3 trillion mentioned above.

The level of carbon prices (or taxes) through time indicates the tightness of policy restraint that is being imposed on carbon emissions along any particular control strategy. Table 9.11 shows the carbon tax under Nordhaus's optimal policy at two points in time, 2010 and 2100. A complete profile of the carbon prices over the next 100 years for the DICE-2007 optimal path is shown in Figure 9.18. These are the tax rates (in constant 2005 US dollar terms) which, if globally imposed on each ton of carbon emissions, would generate the economically optimal path. By construction, along an optimal path the marginal costs of abatement are equal to the marginal benefits of carbon abatement, with each being equal to the carbon tax rate. From inspection of the profile in Figure 9.18, it can be seen that the optimal carbon price follows a gradual rising path, from an initial value of about $27 per ton of carbon to approximately $200 in 2100.[51] The trajectory of optimal carbon prices rises sharply over time as a result of rising marginal damages and the need for increasingly tight restraints.[52] To what level would the carbon price need to rise eventually? This limit is determined by the cost at which a zero-carbon backstop technology becomes available in sufficient quantity to replace carbon-based fuel in all uses. It is evident from Figure 9.18 that such a price is likely to exceed $ 200/tC.

Two points are worth making here. First, while $200/tC is a large tax rate, by 2100 or thereabouts under an optimal mitigation strategy the intensity of control would be such that carbon emission flows will have been driven down to low levels, and so *total* carbon tax payments need not be large. Second, if a backstop technology were to become available

[50] A trillion is a million million (the number 1 followed by 12 zeros). Nordhaus's dollar values are constant price 2005 US dollar equivalents, aggregated over countries using purchasing power parity exchange rates. To get an idea of magnitudes, $3 trillion is around 0.15% of discounted world income.

[51] To convert these carbon prices into their equivalents for carbon dioxide, divide the carbon price by a factor of 3.67.

[52] Table 9.11 and Figure 9.18 also illustrate a number of other mitigation strategies and their associated carbon tax profiles. We shall discuss these below.

Table 9.11 Results of DICE-2007 simulations

	Difference From Base								
	Objective Function	Abatement Plus Damages	PV Climate Damages	PV Abatement Costs	NPV Abatement Costs Plus	Carbon Tax		Global Temperature Change	
						2010	2100	2100	2200
Run	(Trillions of 2005 U.S. $)					2005 U.S. $ per Ton of Carbon		(°C from 1900)	
No controls	0.0	0.0	22.6	0.0	22.6	0.0	1.0	3.1	5.3
Optimal	3.4	3.1	17.3	2.2	19.5	33.8	202.4	2.6	3.5
Concentration limits									
Limit to 1.5 × CO_2	−14.9	−14.6	10.0	27.2	37.2	189.7	761.2	1.6	1.8
Limit to 2.0 × CO_2	2.9	2.7	16.0	4.0	19.9	39.6	445.5	2.5	2.8
Limit to 2.5 × CO_2	3.4	3.1	17.3	2.2	19.5	37.1	202.4	2.6	3.5
Temperature limits									
Limit to 1.5°C	−14.7	−14.4	10.0	27.1	37.0	140.8	899.1	1.5	1.5
Limit to 2.0°C	−1.6	−1.8	13.1	11.3	24.4	60.2	863.4	2.0	2.0
Limit to 2.5°C	2.3	2.0	15.3	5.3	20.6	42.2	539.5	2.4	2.5
Limit to 3.0°C	3.2	3.0	16.7	2.9	19.6	37.9	256.7	2.6	3.0
Kyoto Protocol									
Kyoto with US	0.7	0.6	21.4	0.6	22.0	16.2	11.3	2.9	5.2
Kyoto without US	0.2	0.1	22.4	0.1	22.5	1.2	1.0	3.1	5.3
Kyoto strenthened	1.0	0.7	16.0	5.9	21.9	36.2	321.8	2.4	3.3
Low-cost backstop	17.2	17.2	4.9	0.5	5.4	4.9	4.1	0.9	0.8

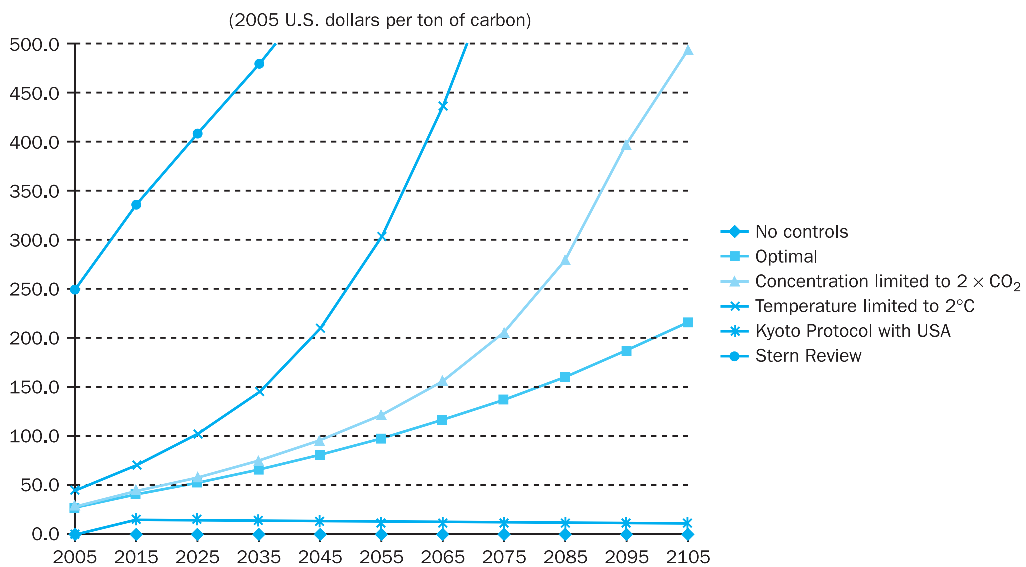


Figure 9.18 The level of carbon prices (or taxes) through time for various mitigation strategies

at both low cost and in large volume in the not too distant future, there would be enormous gains in net economic welfare. The whole carbon tax profile would be shifted downwards in response to the reduced costs of abatement. As a result, there are potentially very large returns to investment in R&D in 'promising' backstops, perhaps including nuclear fusion technology.

2. *Safe minimum standard (precautionary) approaches*

The large uncertainties which exist in climate change modelling regarding the damages that climate change could bring about lead many to conclude that mitigation policy should be based on a precautionary principle. In essence, this would entail that some 'safe' threshold level of allowable climate change is imposed as a constraint on admissible policy choices. The precautionary principle and the safe minimum standard as policy goal are discussed in Chapter 13 where we look at policy in the face of imperfect knowledge about the future. Support for a safe minimum standard approach in the climate change context has grown in recent years for two main reasons. First, the science increasingly points to non-linearities in the dose–response function linking temperature change to induced damages, with damages rising at increasingly large marginal rates at higher levels of global mean temperatures, and possibly discontinuously. Secondly, positive feedbacks in the linkage between GHG concentration rates and temperature responses are increasingly likely to kick in as atmospheric GHG concentrations rise, so that the climate sensitivity coefficients rise endogenously.

Projected climate changes under business-as-usual scenarios encompass temperature changes far outside the range of historical experience. Partly for this reason, the current state of scientific knowledge does not allow us to estimate with any great confidence at what temperature or GHG concentration levels these non-linearities or feedback effects will kick in. However, an increasingly large cluster of expert opinion is now evident which suggests that we are increasingly likely to enter 'dangerous but unknown states' once global mean temperature increases exceed 2°C or GHG concentrations double their pre-industrial levels.

Because of such concerns, it is now commonplace in climate change modelling to analyse policy choices that would be consistent with not allowing GHG concentration rates to exceed some predetermined maximum level, or that do not allow temperature increases to exceed some predetermined maximum level. In the context of maximum allowable GHG concentration rates, once a particular maximum concentration limit has been set, analysts can reason backwards to find out what emission path (or paths) is consistent with that target. Policy instruments can then be set accordingly. Much of what is to be found in IPCC 2007 AR4 is framed in this way.

Working in terms of maximum allowable GHG concentrations or maximum allowable temperature increases bear a close relationship to one another. This is so because, for any particular level of climate sensitivity, it is possible to deduce an equilibrium temperature from a stabilised GHG concentration rate or to deduce a stabilised GHG concentration rate from an equilibrium temperature. This mapping is illustrated in Figure 9.19. Moving from left to right corresponds to increasingly large stabilised (or equilibrium) GHG concentrations that result from successively less stringent mitigation strategies. The variably shaded bands indicate ranges of stabilised GHG concentrations for the IPCC 2007 'stabilisation scenario categories' I to VI. For each particular estimate of climate sensitivity, one can map the stabilised GHG concentration rate to its associated equilibrium global mean temperature change above pre-industrial. The black line in the middle of the

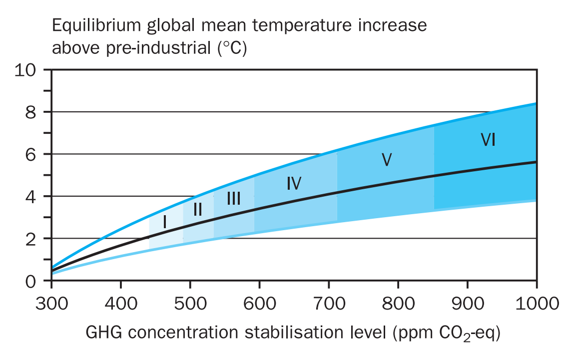


Figure 9.19 Equilibrium global mean temperature increase above pre-industrial (°C)
Source: Figure SPM.8: WGIII SPM. The data are drawn from AR4 WGI, Chapter 10.8

shaded area uses the IPCC 'best estimate' climate sensitivity of 3°C. The upper line of the shaded area gives the relationship for an upper bound estimate of climate sensitivity (4.5°C) and the lower line of the shaded area for a lower bound estimate of climate sensitivity (of 2°C).

Some of the results reported in Nordhaus (2008) allow us to observe how the costs and benefits of an 'optimal' mitigation policy might compare with those that follow a more precautionary approach. Interestingly, Nordhaus finds that 'if concentration or temperature limits are added to the economic optimum, the additional cost is relatively modest for all but the most ambitious targets'. Some indication of these magnitudes can be found by looking at Table 9.11 and Figure 9.18 again. For example, the table shows that imposing a constraint in which CO_2 concentrations are limited to a doubling of pre-industrial levels (labelled as 'Limit to $2.0 \times CO_2$') has an additional present-value cost over that obtained in the optimum of \$0.4 trillion (19.9 – 19.5). A limit to $2.5 \times CO_2$ has no additional cost; such a policy is more or less identical to Nordhaus's optimal strategy in benefit-cost terms. It can also be seen that limiting global temperature increases to 2.5°C has an additional present-value cost of \$1.1 trillion (20.6 – 19.5) over the optimum. These similarities also apply to the carbon tax rates over the first few decades of control, at least for policy interventions that do not impose very stringent stabilisation limits. However, it is clear from Figure 9.18 that in the longer term even quite small tightening of precautionary upper limits on temperature or concentrations results in sharply rising carbon prices. Compare for example, the carbon prices under Nordhaus's optimal policy with those of the prices under a CO_2-doubling limit or a 2°C limit. It is evident that for the various climatic-limit cases, there are steeper increases in the carbon price depending upon the precise target chosen.

This is not surprising. Nordhaus's optimal policy allows an overshooting of temperature and GHG concentration before their steady state equilibria are achieved. Discounting at 'conventional' rates makes it optimal in a present value sense to allow such an overshooting, as the costs of strict control are deferred until later in time. This illustrates the key trade-off with which policy-makers must contend. Standard optimising policies typically ramp up gradually to tight control, thereby moving costs until later when they are less significant in present-value terms. But in so doing, climate change forcings tend to be pushed to greater extremes. If we are worried about the consequences of imposing such forcings, we should be willing to pay a precautionary (or insurance) premium to avoid entering into such ranges.

The 2007 Stern Review report is far more deeply rooted than is Nordhaus in a precautionary approach.[53] In Chapter 3, Box 3.1, we looked at aspects of the recent Stern Review report on the economics of climate change, prepared for the UK government. Box 13.1 in Chapter 13 looks at the way that Stern handles the uncertainty that attends the climate change problem. We note there that although the Stern report is largely in terms of an analysis cast in the expected utility framework, the arguments actually used to arrive at its main recommendation regarding the proper target for mitigation policy have much in common with the Safe Minimum Standard approach to dealing with uncertainty.

9.5.7 International cooperation in climate change policy to date: the Kyoto Protocol

Attempts to secure internationally coordinated reductions in greenhouse gas emissions have taken place largely through a series of international conventions organised under the auspices of the United Nations. At the 1992 'Earth Summit' in Rio de Janeiro, the Framework Convention on Climate Change (FCCC) was adopted, requiring signatories to conduct national inventories of GHG emissions and to submit action plans for controlling emissions. By 1995, parties to the FCCC had established two significant principles in the first instance, emissions reductions would initially only be required of industrialised countries; second, those countries would need to reduce emissions to below 1990 levels.

[53] The Stern Review (2006) is available in full online at http://www.hm-treasury.gov.uk/sternreview_index.htm

The 1997 Kyoto Protocol constitutes the first substantial agreement to set country-specific GHG emissions limits and a timetable for their attainment. To come into force and be binding on all signatories, the Protocol would need to be ratified by at least 55 countries, responsible for at least 55% of 1990 CO_2 emissions of FCCC 'Annex 1' nations. The key objective set by the Protocol was to cut combined emissions of five principal GHGs from industrialised countries by 5% relative to 1990 levels by the period 2008–2012.[54] The Protocol did not set any binding commitments on developing countries.

Since 1997, there have been annual meetings of the parties that signed the Kyoto Protocol. Initially, those meetings were largely concerned with the institutional structures and mechanisms and 'rules of the game' required to implement the protocol, such as how emissions and reductions are to be measured, the extent to which CO_2 absorbed by sinks will be counted towards Kyoto targets, and compliance mechanisms. The administrative structures for some of the flexibility mechanisms, discussed in the next section, were agreed.

The twin conditions required for the Protocol to become operational were met in early 2005. So while the Kyoto Protocol came into force at that time, it did so without the participation of the USA, thereby significantly weakening its potential impact. The first phase of the Kyoto Protocol will end in 2012. Recent meetings of the parties have been concerned with making preparations for its second phase; at the time of writing (May 2010), there is little indication of what kind of arrangements will be agreed for the second phase.

9.5.7.1 The Kyoto Protocol's flexibility mechanisms

The Kyoto Protocol is notable for its provision of several so-called flexibility mechanisms. By generating incentives for control to take place in sources that have the lowest abatement costs, they create the potential for greatly reducing the total cost of attaining any given overall policy target.

- *Emissions Trading* The principle of emissions trading among Annex 1 countries (those industrialised countries for which emissions limits were established) allows countries in which emissions are below their allowed targets to sell 'credits' to other nations, which can add these to their allowed targets.
- *Banking* Emissions targets do not have to be met every year, only on average over the period 2008–2012. Moreover, emissions reductions above Kyoto targets attained in the years 2008–2012 can be banked for credit in the following control period. This provision allows economies flexibility in the timing of their abatement programmes (thereby reducing overall abatement costs), while giving countries incentives to act early.
- *Joint Implementation (JI)* Joint Implementation allows for bilateral bargains among Annex 1 countries, whereby one country can obtain 'Emissions Reduction Units' for undertaking in another country projects that reduce net emissions, provided that the reduction is additional to what would have taken place anyway. Clearly, if the cost of reducing net emissions is lower abroad than at home, this provision will contribute to cost-efficiency. A key issue here is the problem of identifying which projects genuinely are additional ones.
- *Clean Development Mechanism (CDM)* The Clean Development Mechanism creates the potential for further efficiency gains in policy implementation. These arise where projects that reduce emissions in developing countries are less costly than equal-sized reductions in Annex 1 nations. By funding such projects, Annex 1 countries can gain emissions credits to offset against their abatement obligations. Effectively, the CDM generalises the JI provision to a global basis. The CDM applies to sequestration schemes (such as forestry programmes) as well as emissions reductions. Again, there is concern that some trades under CDM could be spurious in that the project would have taken place

[54] The agreement also specified the amount each industrialised nation must contribute towards the overall target. Most country-specific targets mandated reductions of between 6% and 8%, but a few countries were allowed to maintain emissions unchanged over the specified period or were allowed emissions increases because of special circumstances.

anyway (and so should not create offsets elsewhere). As with JI, institutional arrangements are necessary to validate project additionality.

Kyoto's flexibility mechanisms appear to offer very large prospects of reductions in overall emissions abatement costs. Studies carried out during the 1990s found median marginal abatement costs in developed economies to be of the order of \$200 per tonne of carbon. Barrett (1998) argued that with emissions being uncontrolled in the non-Annex 1 countries, marginal abatement costs there are effectively zero. On this basis, he suggests that cost savings from the Clean Development Mechanism alone could be of the order of \$200/tC at the margin.

9.5.7.2 An appraisal of the Kyoto Protocol

The game-theoretic literature that we discussed earlier in this chapter does not point to a high likelihood of efficient outcomes when it comes to attempts to control climate change. Global climate change fits well with the Prisoner's Dilemma game template, in which abatement inactivity is a dominant strategy. International cooperation is made difficult by each of the following characteristics:

1. Negotiation concerns benefits that are largely public (rather than private) good in nature.
2. The number of affected countries is very large.
3. For many countries at least, nation-specific benefits are small relative to transnational benefits.
4. The benefits of climate change control are highly uncertain and very unevenly distributed.
5. Abatement or mitigation costs to many individual countries are perceived to be high.[55]

These considerations suggest that agreements which involve widespread commitments to undertake substantial amounts of climate change control will be difficult to secure and implement. Of most concern at the time of writing is that the USA has not ratified the Kyoto Protocol, and that developing countries have not been brought into the set of controlled countries. As things stand, the Kyoto Protocol is not a complete treaty since it does not support full participation, it lacks internationally enforceable compliance mechanisms, and it lacks incentive mechanisms that would make the Protocol self-enforcing.

One might argue that it is churlish to be critical of Kyoto on the basis of a comparison with a 'first-best' policy. A realistic interpretation of the Kyoto Protocol must recognise that it is just a limited first step on the road to a more comprehensive and efficient IEA on climate change. It has generated a number of valuable institutional mechanisms, and it has done a powerful job of creating political capital in support of cooperative action. Nevertheless, some authors – while recognising this 'first-step' aspect of Kyoto – argue that fundamental changes need to be made to improve its performance.

Barrett has argued that it was wrong to follow the Montreal Protocol's (on ozone-depleting substances) primary focus on emissions targets and timetables, as these were almost certain to lead to incomplete participation and limited abatement effort. In his view, a better way forward would have been a technology-centred approach: international co-operation on mitigating climate change should initially focus on broad cooperation in the research and development of new environmentally friendly technologies, moving the world away from its dependence on carbon-based fuels, and on negotiating protocols establishing standards for the adoption of these technologies. This would generate a network externality effect: as new technologies become more widely adopted, costs of those technologies fall and their market size expands while markets for harmful technologies decrease. At some threshold diffusion rate, it would be in the interests of all countries to ratify. Such a scheme would be relatively easy to administer, could be supported by trade restrictions on imports based on harmful technologies, and could achieve desirable distributional goals by the creation of an international fund to foster the spread of new technologies in developing countries.

It is interesting to note with regard to Barrett's criticism of the Kyoto Protocol that the flexibility

[55] While several of these features are also true for control of CFC and other ozone-depleting emissions, it is noteworthy that control costs are very much lower and benefits more evenly spread in that case. These factors possibly explain the relatively successful efforts on that front.

mechanisms are really an add-on, which many were reluctant to allow. Economists criticise any lack of such flexibility, but some who opposed it were motivated by the argument that the main thing to secure was the driving of technical change in a non-carbon direction, and the idea that the nature of technical change would be driven by what took place in the USA and Europe. The point was to force innovation in those places so that it could be diffused to the rest of the world. The objection to flexibility was that letting the UK, for example, meet its obligations by financing reforestation in Brazil might be 'least cost' but it would not focus UK minds on decarbonising electricity supply.

Nordhaus (2008) and Stern (2007) make rather different criticisms of the Kyoto Protocol. Unlike Barrett who argues for targeting technology, Nordhaus and Stern both accept the Kyoto approach on targeting emissions. But they both object to the basis on which Kyoto Phase 1 targets were set. They argue that its emissions targets and timetables are arbitrary, having no explicit grounding in either economic or environmental objectives. In particular, its approach of freezing emissions for a subgroup of countries is not related to a particular goal for GHG concentrations, global mean temperature, or damages. While Phase 1 provisions do pass a minimal test of having positive net benefits, those net benefits are very small relative to ones obtained under other policies.[56]

9.5.7.3 After Kyoto?

The first 'commitment period' of the Kyoto Protocol will expire at the end of 2012. A Conference of the Parties was held in December 2009 in Copenhagen, Denmark, in the lead-up to which it had been hoped that a treaty to succeed the Kyoto Protocol would be adopted.[57] At the time of preparing final edits to this chapter, the 2009 United Nations Climate Change Conference, commonly known as the Copenhagen Summit, had just taken place. In spite of many hopes (and some expectations) that ambitious and legally binding targets for greenhouse gas (GHG) emissions reductions would be agreed, the summit failed to live up to those hopes and expectations.[58] The conference was plagued by negotiating deadlock and its formal outcome, the 'Copenhagen Accord', is not legally enforceable. In effect, substantive and binding commitments were deferred until a later date, the next opportunity being at the 2010 UN climate change conference in Mexico. However, some progress has been made on two fronts. First, there seems to be some degree of convergence towards the principle that climate agreements must aim to keep average global temperature rises to at most 2° Celsius above pre-industrial levels. This figure – although being higher than some would like to see – corresponds to one that has been widely mentioned in the scientific literature as a likely threshold point beyond which damages will rise sharply and discontinuously. Second, commitments were made by developed countries to provide funds (US$ 30 billion for the period 2010–2012, rising to US$ 100 billion per year by 2020), to the developing world to support mitigation of, and adaptation to, climate change.

In the months leading up to the Copenhagen meeting, many countries had set out their own emission reduction targets, although in most cases these commitments were conditional on what others would do.[59]

[56] For example, Nordhaus calculates that the current Kyoto Protocol (without the United States) has net benefits of around $0.15 trillion, compared with $3.4 trillion for his optimal policy.

[57] Readers should refer to the UNFCCC website (http://unfccc.int) for official contributions to this process, and to keep abreast of changes that occur after publication of this textbook.

[58] For example, earlier proposals that would have aimed to limit temperature rises to 1.5°C and cut CO_2 emissions by 80% by 2050 were dropped, and no legally binding emissions reduction schedules for the participating countries were agreed.

[59] Some examples:

Australia: To cut carbon dioxide emissions by 25% below 2000 levels by 2020 if the world agrees to an ambitious global deal to stabilise levels of CO2e to 450 ppm or lower; to cut carbon dioxide emissions by 15% below 2000 levels by 2020 if there is an agreement where major developing economies commit to substantially restrain emissions and advanced economies take on commitments comparable to Australia; and to cut carbon dioxide emissions by 5% below 2000 levels by 2020 unconditionally. These proposals are equivalent to an emissions cut of 24%, 14%, and 4% below 1990 levels by 2020 respectively.

China: To cut emissions intensity by between 40% and 45% below 2005 levels by 2020. (Conditional pledge.)

European Union: To cut GHG emissions by 30% below 1990 levels by 2020 if an international agreement is reached committing other developed countries and the more advanced developing nations to comparable emission reductions; to cut greenhouse gas emissions by 20% below 1990 levels by 2020 unconditionally.

United States of America: To cut greenhouse gas emissions by 17% below 2005 levels by 2020, 42% by 2030 and 83% by 2050. This is equivalent to 1.3% below 1990 levels by 2020, 31% by 2030 and 80% by 2050. (Conditional pledge.)

In the absence of specific agreements on emission reductions, the Copenhagen Accord asks countries to submit emissions targets by the end of January 2010, but at the time of writing it is not clear what will actually come out of this request. It is also not yet (and it may never be) clear why Copenhagen arrived at an outcome that disappointed so many. We have already seen just how formidable is the achievement of self-enforcing treaties that engender large changes in behaviour and at the same time generate widespread support among a large number of parties. Various attempts have been made to interpret and explain the negotiation processes and outcomes, and allocate blame for perceived failures to reach substantive agreement. But at the root of the problem seems to be fundamental differences between richer and poorer countries about the magnitudes of emissions reductions that should be expected of the developed and developing worlds, and how the burdens of achieving the reductions should be distributed among nations. This set of issues was side-stepped by Kyoto, but hopes that it could be easily resolved in Copenhagen have been dashed.

Learning outcomes

Near the start of this chapter, it was stated that our objective was to provide the reader with the means to answer several fundamental questions, and to apply those answers to specific problems. We here repeat those questions, and provide (very briefly) some preliminary answers implied by the analysis of this chapter.

1. In which ways do international environmental problems differ from purely national (or sub-national) problems? The main point here is that these problems involve people living (or yet to be born) in more than one country. There are spillovers or externalities that cross national boundaries. This poses difficulties because political sovereignty is typically given to the nation state.
2. What additional issues are brought into contention by virtue of an environmental problem being 'international'? Given the previous comments, it is evident that outcomes are likely to be inefficient or sub-optimal unless nations cooperate and policy is coordinated.
3. What insights does the body of knowledge known as game theory bring to our understanding of international environmental policy? Game theory provides a framework for analysing behaviour where the pay-offs to choices are strategically interdependent (that is, where the pay-off to one player from a decision depends upon the choices of all other players in the 'game').
4. What determines the degree to which cooperation takes place between countries and policy is coordinated? What conditions favour (or discourage) the likelihood and extent of cooperation between countries? We have brought together, in Box 9.1, a statement of these conditions. You should re-read that now. Of particular importance seem to be reciprocity and repetition. If parties interact continuously – either having to make repeatedly a particular choice, or negotiate with each other repeatedly but about different things, the chances of effective cooperation are greatly enhanced.
5. Why is cooperation typically a gradual, dynamic process, with agreements often being embodied in treaties or conventions that are general frameworks of agreed principles, but in which subsequent negotiation processes determine the extent to which cooperation is taken? It is often best to enter a cooperative process by making small commitments; cooperation may then be stepped up gradually as parties are observed to stick to agreed principles, and as new knowledge arrives.
6. Is it possible to use such conditions to explain how far efficient cooperation has gone concerning lower-atmosphere ozone and greenhouse gas pollution? The answer seems to be that the appropriate theory can be, and has been, used with considerable success.
7. Most importantly, these insights will help in the design of future cooperative processes and ventures. Well-designed bargaining mechanisms can, in some circumstances, generate substantial mutual net benefits to participants (relative to non-cooperative outcomes).

Further reading

Game theory

Good discussions of game theory at an elementary to intermediate level, but not explicitly linked to issues of environmental policy, are available in Varian (1987), chapters 27, 31 and 32 and Rasmusen (2001). Mäler (1990) treats the topic at a slightly more advanced level and with an environmental economics focus. Barrett (1990, 1994a, 2003) explores cooperative and non-cooperative outcomes for a range of types of spillover, develops the concept of self-enforcing international agreements, and considers a range of IEPs in the light of game theory concepts. Other good sources of information on IEAs include Keohane *et al.* (1993), and Finus (2002, 2004). Hoel (1989) demonstrates the worrying result that 'unselfish' unilateral action can result in outcomes that lead to greater levels of emission than in its absence. Dasgupta (1990) shows that cooperation need not require an outside agency to enforce agreements and that such cooperation could be sustained over time by means of norms of conduct. Victor *et al.* (1998) discuss the effectiveness of international commitments. Evolutionary game theory – which has important applications in ecology and ecological economics – is discussed in Axelrod (1984), a beautiful and easy-to-read classic, and the more difficult text by Gintis (2000b).

Uncertainty and international environmental agreements

The results about IEAs summarised in the chapter were obtained in the main from models in which the benefits and costs of emissions abatement are known with certainty. There is a rapidly growing literature analysing how uncertainty affects the incentives for countries to join an IEA. That uncertainty could be about control costs and/or damage costs, or about the possibility of resolving that uncertainty in the future. Important recent contributions have been made by Ulph (2004) and Kolstad (2007). A good discussion of post-Kyoto architectures (for which uncertainty is of the essence) is Aldy and Stavins (2007).

International coordination of policy and the use of tradable permits

Grubb (1989a) provides an excellent critical survey of the various initial allocation options for marketable permit systems to achieve internationally agreed pollution control targets. Other analyses are found in Hahn and Hester (1989b), Bertram *et al.* (1989) and Tietenberg (1984, 1990). These sources also discuss the distributional consequences of various alternative methods of allocating permits between countries. See also WR (1996), chapter 14. For the use of tradable permits for GHGs, see 'Global climate change' below.

Acid rain

An extensive account of issues concerning acid rain is available in the document *Acid Rain.doc* available on the Companions Website. The scientific basis is well described in Kemp (1990) and a definitive study is to be found in NAPAP (1990). A good analysis of the acid rain issue is to be found in Adams and Page (1985). *World Resources*, published every two years, provides regular updates of the scientific evidence and economic assessments of the damages caused. Good economic analyses may be found in Feldman and Raufer (1982) and Tietenberg (1989). Two articles in the summer 1998 issue of the *Journal of Economic Perspectives* – Schmalensee *et al.* (1998) and Stavins (1998) – give authoritative appraisals of the United States SO_2 emissions trading programme.

Ozone depletion

Kemp (1990), WMO (1991) and French (1990) describe the scientific basis of ozone depletion. *World Resources* provides regular updates. An excellent economic analysis is in Bailey (1982). See also Office of Air and Radiation *et al.* (1995).

Global climate change

The most comprehensive and up-to-date surveys of climate change science, economics and policy is found in the three most recent IPCC reports: IPCC(1), IPCC(2) and IPCC(3), all published in

2007. The full versions of each of these reports an integrated summary of all three (known as 'The AR4 Synthesis Report'), and a *Summary for Policy Makers* and a *Technical Summary* for each of the three individual full reports – together with a variety of background papers – can be downloaded from the IPCC website at www.ipcc.ch. Important policy oriented economics-based studies of climate change are found in the Stern Review (2006) and Nordhaus (2008). The Stern Review attracted much criticism from other climate change modellers. The Weitzman, Nordhaus and Mendelsohn reviews are devastating. Nordhaus's critique of Stern can be found in his 2008 book.

A series of papers that together constitute a comprehensive general survey of the economics of global climate change is found in Toman (2001). Other general surveys include Rao (2000) and DeCanio *et al.* (2000), available online at http://www.pewclimate.org/global-warming-in-depth/all_reports/new_directions. The website of the Pew Centre on Climate Change (at http://www.pewclimate.org/) contains a wealth of freely downloadable papers and conference discussions among eminent thinkers in the field. One good example of the latter is the Pew Centre's 2009 workshop on assessing the benefits of avoided climate change, online at http://www.pewclimate.org/benefitsworkshop-March09.

Hall and Howarth (2001) examine the long-term economics of climate change. An early presentation of the scientific basis for the greenhouse effect, written from the perspective of an environmental economist, is given in Cline (1991). A good summary of the scientific basis for possible abrupt, fast and large scale changes in climate is given in UK Parliamentary Office for Science and Technology (2005), available online at http://www.geos.ed.ac.uk/ccs/Post245.pdf. For information on the warming contribution of different GHGs, see Houghton *et al*. (1990), Grubb (1989b), Lashof and Ahuja (1990), Nordhaus (1991a), and the 2007 IPCC Reports. The best general accounts of emissions forecasts and/or scenarios are to be found in the various 2007 IPCC Reports. Earlier coverage can be found in Reilly *et al*. (1987), IPCC (1992, 1994), IPCC(1, 2 and 3) (2001), IEA (1995), World Energy Council (1993) and in report DOE/EIA-0484 (95) from the Energy Information Administration (1995). Further information on impacts and damages is given in Schneider (1989), Cline (1989, 1991), Nordhaus (1990a, b), EPA (1988, 1989), IPCC (1995a), Hansen *et al.* (1988), IPCC(2) (2001), and Mendelsohn (2001). Common (1989), Hansen (1990) Nordhaus (1991a, 2008) and Stern (2007) discuss the uncertainties involved in damage estimation. Smith *et al.* (2009) show how the IPCC's 'reasons for concern' about climate change damage increased substantially between its third (2001) and fourth (2007) assessment reports.

Early studies concerning the costs of CO_2 (and other GHG) abatement can be found in Department of Energy (1989) for the UK, Manne and Richels (1989), IPCC (1995b), Nordhaus (1990a, 1990b, 1991a, 1991b), Barker (1990), Jorgensen and Wilcoxen (1990a, 1990b, 1990c), Anderson and Bird (1990a, 1990b), Cline (1991), Edmonds and Barns (1990a, b), Edmonds and Reilly (1985) and Mintzer (1987). Early CGE-based simulations can be found in Whalley and Wigle (1989, 1990, 1991) and Burniaux *et al.* (1991a, 1991b, 1992). More recent studies of GHG abatement costs are to be found in White *et al.* (2001) and Hall and Howarth (2001). Green (2000) considers scale-related problems in estimating the costs of CO_2 mitigation policies. Good examples of attempts to use economic analysis to develop policy options and/or GHG policy targets are Nordhaus (2001, 2008), Goulder and Mathai (2000), which examines optimal abatement in the presence of induced technological change, and Stern (2007).

Policy instruments and global warming are discussed in Opschoor and Vos (1989) and Pearce (1991b). and in many of the other references listed in this section. Early discussions of appropriate responses under uncertainty include Barbier and Pearce (1990) and Hansen (1990). More recent analyses can be found in Chichilnisky and Heal (2000), Weyant (2000, available online at http://www.pewclimate.org/docUploads/econ_introduction.pdf), Hohmeyer and Rennings (1999), Fankhauser *et al.* (1999), Petsonk *et al.* (1998, available online at http://www.pewclimate.org/upublications/report/market-mechanisms-global-climate-change, and several documents that can be found by searching the Pew Centre website. Tietenberg (1984, 1990) discusses tradable emissions permits; Grubb (1989a) argues that internationally tradable permits represent the best approach for international action towards the greenhouse effect. Edmonds *et al.* (1999) examine the effects of international emissions trading on abatement costs (available online at http://www.pewclimate.org/global-warming-in-depth/all_reports/international_emissions/). Adger *et al.* (1997) consider climate change mitigation and European land use policies. Barrett (1998) provides an interesting discussion of the 'political economy' of the Kyoto Protocol. Information about the evolving markets for GHG emissions trading may be obtained at the UK Emissions Trading Group website at www.uketg.com and the European Union Emissions Trading System (EU ETS) at http://ec.europa.eu/environment/climat/emission/index_en.htm. An assessment of the EUETS is found in Ellerman and Joskow (2008), available online at http://www.pewclimate.org/eu-ets, with a briefer Pew Centre review at http://www.pewclimate.org/EU-ETS-history.

Optimal policy towards the greenhouse effect given uncertainty and learning is analysed in Kolstad (1996). Use of performance bonds as a way of dealing with scientific uncertainty within a precautionary principle framework is discussed in Costanza and Perrings (1990) and Costanza and Cornwell (1992). The latter named authors apply the idea to issues of global climate change. Boscolo and Vincent (2000) examine the prospects for using environmental performance bonds as an instrument for reducing the negative externalities associated with logging practices in tropical rain forests, and find these prospects to be promising.

Discussion questions

1. Discuss the proposition that marketable emissions permits are more appropriate than emissions taxes for controlling regional and global pollutants because of the much lower transfer costs associated with the former instrument.

2. Consider the following extracts from an article in *The Independent* newspaper (28 March 1995) by the economist Frances Cairncross:

   > Work by William Cline, a scrupulous and scientifically literate American economist, suggests that the benefits of taking action do not overtake the costs until about 2150. And Mr Cline sees global warming largely in terms of costs. Yet it is inconceivable that a change of such complexity will not bring gains . . . as well as losses.
   >
   > Given the difficulties of doing something about climate change, should we try? Some measures are certainly worth taking because they make sense in their own right. . . . Removing such [energy] subsidies would make the economy work more efficiently and benefit the environment, too.
   >
   > Indeed, wise governments should go further, and deliberately shift the tax burden away from earning and saving . . . towards energy consumption.
   >
   > Beyond that, governments should do little. The most rational course is to adapt to climate change, when it happens. . . . Adaptation is especially appropriate for poor countries once they have taken all the low-cost and no-cost measures they can find. Given the scarcity of capital, it makes good sense for them to delay investing in expensive ways to curb carbon dioxide output. Future economic growth is likely to make them rich enough to offset those effects of climate change that cannot be prevented.

   Provide a critical assessment of these arguments.

3. Compare and contrast the cost-effectiveness of
   (a) a sulphur dioxide emission tax;
   (b) a sulphur dioxide emission tax levied at the same rate as in (a), together with an arrangement by which emissions tax revenues are used to subsidise capital equipment designed to 'scrub' sulphur from industrial and power-generation emissions.

4. The text mentioned that forests, agricultural lands and other terrestrial ecosystems offer significant carbon-reduction potential. Choose one country, and consider the ways in which this potential might be realised. What are the limits to the extent to which these methods could be used?

Problems

1. The world consists of two countries, X which is poor and Y which is rich. The total benefits (B) and total costs (C) of emissions abatement (A) are given by the functions

$$B_X = 8(A_X + A_Y), \quad B_Y = 5(A_X + A_Y),$$
$$C_X = 10 + 2A_X + 0.5A_X^2 \text{ and}$$
$$C_Y = 10 + 2A_Y + 0.5A_Y^2$$

where the subscripts are used to denote the country, X or Y, in which abatement takes place.

(a) Obtain the non-cooperative equilibrium levels of abatement for X and Y.
(b) Obtain the cooperative equilibrium levels of abatement for X and Y.
(c) Calculate the utility levels enjoyed by X and by Y in the non-cooperative and cooperative solutions. Does the cooperative solution deliver Pareto improvements for each country, or would one have to give a side-payment to the other to obtain Pareto improvements for each with cooperation?
(d) Obtain the privately optimising level of abatement for X, given that Y decides to emit at the level of emissions that Y would emit in the cooperative equilibrium. (You should find that the answer to (d) above is that X does the same amount of abatement that she would have done in the non-cooperative case. What property or properties of the cost and benefit function used in this example cause this particular result?)
(e) Suppose that Y acts as a 'swing abater', doing whatever (non-negative) amount of abatement is required to make the combined world abatement equal to the combined total under a full cooperative solution. How much abatement is undertaken in the two countries?

2. Refer to Figure 9.10. Show how the relative slopes of the MB and MC functions, and the number of countries, N, determine

(a) the magnitude of the efficiency gain from full cooperation, and
(b) the amount by which the cooperative level of abatement exceeds the non-cooperative (Nash) abatement level.

What conclusions can be drawn from your results?

CHAPTER 10 Trade and the environment

Just between you and me, shouldn't the World Bank be encouraging MORE migration of the dirty industries to the LDCs [Less Developed Countries]?

Extract from memo allegedly written by Lawrence Summers, then Chief Economist at the World Bank, in 1991

Learning objectives

After reading this chapter you should be able to answer the following questions

- What are the classical theorems of international trade and the implications of extending them to environmental resources?
- When does trade liberalisation damage the environment and if so does it matter?
- Do countries competitively reduce environmental standards and if so for what purpose?
- Does banning trade in products made from endangered species necessarily protect those species?
- Does adherence to the General Agreement on Tariffs and Trade unduly constrain attempts to tackle environmental problems?
- Does the empirical evidence suggest that differences in the stringency of environmental regulations are a major determinant of trade flows?

Introduction

Concerns expressed by environmentalists about international trade frequently include the following: Will gains from trade liberalisation be outweighed by the damage to the environment? Is there an incentive for countries to engage in 'ecological dumping' where one country lowers its emissions standards to gain some competitive advantage? Will footloose industries migrate to 'pollution havens'? Will countries competitively lower environmental standards to generate trade benefits resulting in a 'race to the bottom'? What about the environmental impact of transporting goods internationally?

Businessmen often argue that environmental regulations will result in the loss of jobs, and that anyway, the production of pollution intensive goods will merely be pushed offshore. And politicians of course, are concerned about the resulting unemployment.

Debate about the impact of international trade on the environment grew noisier in the early 1990s, mainly in response to fears about the possible impact of the proposed North American Free Trade Agreement (NAFTA) between Canada, the USA and Mexico. Standing on a populist platform former US presidential candidate Ross Perot claimed that NAFTA would result in a giant sucking sound of jobs flowing to Mexico and recommended that Mexican goods be barred from entering the USA unless they were produced under conditions meeting US standards for environmental protection. The exchange between Daly (1993) and Bhagwati (1993) is between two of the chief protagonists in the debate over NAFTA.

But many would take a different view of things. That free trade improves economic welfare is a cherished belief for most economists. And some – like the former chief economist of the World Bank quoted above – seem positively to welcome the fact that industries might wish to relocate in countries with low standards. For them it is nothing more than a demonstration that differing endowments are the basis of trade. Nor do economists generally accept the argument that countries ought to share the same environmental standards given the existence of large differences in per capita incomes and population densities between countries. Many economists therefore, it seems fair to say, would see no reason for biophysical standards and environmental taxes to be harmonised. And although changes in environmental regulation might result in temporary unemployment, changes in the exchange rate will ultimately restore the trade balance. Calls for trade restrictions made on environmental grounds should be viewed with suspicion as they might be motivated by protectionist concerns.

To complete the picture there is even the view, attributed to Porter (1991), suggesting that governments should set tough environmental standards as a way of inducing domestic producers to innovate new green technologies, thereby securing them a long-term competitive advantage.

The structure of the remainder of the chapter is as follows. We start by reminding ourselves of the traditional theory of trade and attempt to gain insights extending this theory to incorporate environmental resources. We then enquire more closely why particular countries might have an advantage in the production of environmentally intensive commodities. We present a partial equilibrium analysis of the effects of moving from a position of autarky (no trade) to a system of free trade with and without the requisite controls on the production of environmental externalities. We will show that the problem is the absence of proper, i.e. economically efficient, environmental policy rather than trade liberalisation *per se*.

We then present a general equilibrium model to examine the effect of trade liberalisation on welfare. Environmental dumping refers to a situation in which a country employs unduly lenient environmental standards in order to pursue trade-related objectives. We next consider situations in which governments might attempt to manipulate environmental standards in order to benefit national producers. We then develop a model in which jurisdictions compete in order to attract capital and once more ask whether this is likely to result in an erosion of environmental standards.

Since it is relatively easy for trade protection to masquerade as environmental protection, we then look closely at the role played by the World Trade Organisation (WTO) in policing the global trading system. What, according to the WTO, is the difference between justifiable and unjustifiable restrictions on trade enacted in the name of environmental protection? Lastly, we provide a brief overview and critique of attempts to test the proposition that environmental regulations explain trade flows and patterns of foreign direct investment (FDI).

10.1 An environmental extension to traditional trade theory

Traditional trade theory is characterised by a number of famous theorems. These include the Heckscher–Olin theorem of trade, the Stolper–Samuelson theorem, the Rybczinski theorem, and the factor price equalisation theorem. It is not our intention here to provide anything other than a brief outline of each of these theorems. These are covered by any standard textbook on international trade, e.g. Bhagwati and Srinivasan (1983). Instead, like Rauscher (1999) and Steininger (1999) we wish to consider the implications of these theorems when environmental factors of production are brought into the picture alongside capital and labour.

The Heckscher–Ohlin theorem is a fundamental proposition regarding the pattern of trade between two economies. It suggests that trade is determined by differences in factor endowments. A country will export those goods relatively intensive in its abundant factor of production and import those goods relatively intensive in its scarce factor of production. For example, a country having an abundant supply of capital will find it cheaper to manufacture goods whose production is capital intensive. The country is said to have a ‘comparative advantage’ in the production of such goods. Post free trade the consumption of the labour-intensive goods will increase because of an increase in national income and a positive substitution effect whereas the consumption of capital-intensive goods may or may not increase. Taken as a whole, consumers unambiguously reach a higher level of welfare.

The Heckscher–Ohlin theorem rests on a number of assumptions. Countries are assumed to have identical constant returns to scale technologies and identical tastes. There are no impediments to trade and no transport costs. There are two goods, two factors of production available in fixed quantities, and two countries one of which is small and the other one large representing the rest of the world. Factors of production are assumed to be immobile.

We now consider whether and how the Heckscher–Ohlin theorem can be extended to include environmental and natural resources as factors of production. Obviously the exports of many countries reflect their exploitation of particular natural resource endowments. But the observation that countries e.g. with deposits of bauxite should 'specialise' in the production of bauxite ore does not seem especially insightful. Neither does the observation that countries endowed with oceanic resources should specialise in fishing seem worthy of comment. But environmental resources include air, soil and water quality, and the capacity of the environment to assimilate the unwanted by-products of economic activity. From our perspective a country is therefore well endowed with environmental resources when it is sparsely populated and has a higher assimilative capacity.

According to the Heckscher–Ohlin theorem such countries should specialise in the production of environmentally damaging goods. Ultimately, however, a country's endowment of environmental resources is determined by environmental regulation. The stringency of environmental regulation should reflect the population's appetite for environmental quality. Here we should distinguish between the 'true' endowment and the *de facto* endowment resulting from lobbying from special interest groups. Moreover, as Copeland and Taylor (1994) point out, the stringency of environmental policy is endogenous so the growth in income that accompanies trade liberalisation might increase the stringency of environmental policy and therefore reduce the endowment of environmental factors of production.

Differences in the abundance of environmental resources and differences in the appetite of populations for environmental quality make it fairly clear that the harmonisation of environmental standards is inappropriate. There are perfectly legitimate reasons why governments should wish to set different environmental standards.

The Heckscher–Ohlin theorem predicts that tighter environmental regulation at home leads to the increased production of environmentally intensive products abroad (although we shall later go on to qualify this statement). These environmentally intensive products are then imported. The most topical example of this is where a single country or a bloc of countries imposes restrictions on carbon dioxide emissions. According to theory this should result in an increase in carbon dioxide emissions elsewhere, something referred to as 'carbon leakage'. Economists have utilised Computable General Equilibrium (CGE) models to estimate the extent of carbon leakage associated with attempts to reduce carbon emissions by, say, OECD countries or the EU. It is easy to understand why the threat of carbon leakage might prompt policy makers to (a) think about trade restrictions to prevent the import of carbon-intensive commodities and (b) reconsider the wisdom of unilateral measures to reduce carbon dioxide emissions.

Differences in population density, the waste assimilative capacity of countries and national preferences for environmental quality also provide a basis for trade in waste and residuals themselves (in fact such trade is restricted by the Basel convention). Equally, a country with especially stringent environmental standards might export commodities whose production is particularly sensitive to poor environmental quality.

The Heckscher–Ohlin theorem explains why a country is able to increase its income through trade. But this does not mean that free trade will be advantageous for everyone. The Stolper–Samuelson theorem demonstrates that those who supply the scarce factor of production can gain through protection that restricts imports of goods intensive in that factor. More formally, a tariff will increase the income of the factor used intensively in the good that receives protection. The intuitive reason is that the resulting increase in the price of the good increases the derived demand and therefore the price of the intensively employed factor. Unskilled labour in the EU, for example, has every incentive to lobby for protection on labour-intensive goods from outside the EU.

Many environmental resources are not privately owned. But where environmental resources are subject to private property rights (perhaps because these have been created by the government in the form of tradable permits), the Stolper–Samuelson theorem may apply. Expect therefore to see private owners of environmental permits lobbying for trade protection from imports intensive in the relevant environmental resource from countries where its use is uncontrolled.

The Rybczynski theorem states that an increase in the endowment of one factor will reduce the production of goods intensive in the other factor. For example, if there is an increase in labour the production of capital-intensive goods will decline. This theorem relies on the small country assumption that commodity prices remain unchanged when factor supplies increase. A *de facto* increase in environmental resources, caused for example by an increased allocation of environmental permits by an incoming government that cares less for environmental quality, should decrease the production of environmentally less-intensive commodities.

Heckscher–Ohlin suggests that free trade would equalise factor prices since a country would export goods intensive in its abundant factor of production and import goods intensive in its scarce factor of production. This would decrease the derived demand for the scarce factor of production and increase the demand for the abundant factor of production, thereby reducing international differences in factor prices. This was subsequently proved by Samuelson (1948) who demonstrated that the assumptions underlying the Heckscher–Ohlin theorem were indeed sufficient to equalise the price of factors even in the absence of factor mobility. Only one additional assumption is required, namely that the factor endowments are not too different such that countries are not driven to complete specialisation.

Despite the fact that there is in fact lots of capital, and quite a lot of labour, mobility, casual observation suggests that factor price equalisation theorem does not hold. Unskilled workers in the developed world continue to earn far more than equivalent workers in Less Developed Countries. This could be attributed to the rather rigid assumptions upon which the theorem, and indeed the Heckscher–Ohlin theorem, is based e.g. the existence of free trade and the assumption that production technologies are the same everywhere.

The factor price equalisation theorem would seem to suggest that international trade equalises the shadow price of environmental resources – a striking result. The shadow price of environmental resources means environmental tax rates or the price of environmental permits. Note that this does not require tax rate harmonisation or trade in permits. This of course does not mean that environmental quality will be equalised. Note, however, that for more than two goods and more than two factors of production only a weaker version of the Stolper–Samuleson theorem holds.

10.1.1 North–South models of trade and the environment

We will now turn to consider other sources of comparative advantage. So-called North–South models suggest that the source of the poor South's advantage is not that they have abundant environmental resources but that the use of these resources is unregulated by the government. Such models are associated with the seminal paper of Chichilnisky (1984). This gives the South an illusory comparative advantage in the production of environmentally intensive goods. It is customary to refer to the region suffering from absent property rights governing access to the environmental resource as the 'South' and the region in which property rights both exist, and are enforced, as the 'North'. Owing to the mismanagement of environmental resources, trade may exacerbate the environmental problems of the South while the rich North benefits from trade.

10.2 Does free trade harm the environment? A partial equilibrium analysis

The preceding section extended classical theories of trade to include environmental resources. We will now address the twin questions of (a) whether trade is bad for the environment and (b) whether it matters in a partial equilibrium setting in which the production of a single good gives rise to external costs. For the sake of simplicity these external costs are assumed to arise from the production of the good rather than the use of particular production processes. The country is assumed to be small such that it cannot affect its own terms of trade. Factors of production are assumed to be immobile and there is no transboundary pollution.

This analysis presented below follows closely that of Anderson (1992). Assume that the country switches from an initial position of autarky to one of free trade. It turns out that there are four cases to consider: (a) the country already imposes an (allocatively efficient) environmental tax or permit quantity of the correct amount and becomes a net importer, (b) the country already imposes an environmental tax and becomes a net exporter, (c) the country does not impose an environmental tax and becomes a net importer, and (d) the country does not impose an environmental tax and becomes a net exporter.

Liberalising trade in the good whose production harms the environment can be welfare enhancing or not depending upon whether the country already imposes the efficient tax rate on domestic production. To begin with, suppose that it does not. If the good ends up being imported then welfare is definitely improved. This is because real resource costs fall while at the same time domestic pollution levels fall.

In Figure 10.1 the supply curve S represents the domestic supply curve of the commodity whilst the curve S* represents the supply curve including the environmental costs. The curve D represents the domestic demand and the horizontal line P represents the international price of the commodity. Prior to trade liberalisation domestic production and consumption is at point Q. Following free trade domestic output falls to Q_m whereas domestic consumption rises to C_m, the difference representing imports. There is an increase in producer and consumer surplus corresponding to area gef, plus a reduction in environmental costs equal to ghde.

In Figure 10.2 once more S represents the domestic supply curve of the industry, S* represents the supply curve plus the environmental costs and D represents the domestic demand curve. The horizontal line P is now higher, indicating that opening up to trade will result in a higher price for consumers and increased domestic production leading to exports. Production rises from Q to Q_x while consumption falls to C_x. The change in economic surplus is given by eik, although there is an increase in environmental costs of dekm. The relative size of these areas and therefore the overall welfare impact is unclear. Consequently, if a country without environmental taxes ends up being a net exporter then the gains from trade may be more than outweighed by the losses from increased production and the external costs that this generates – a situation of some concern.

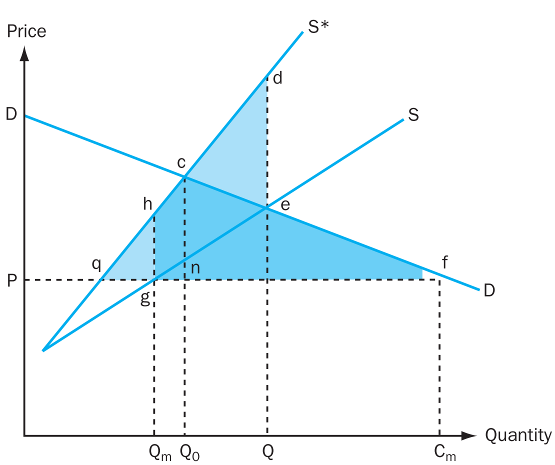


Figure 10.1 The effect of free trade in a commodity whose production is polluting: the importables case

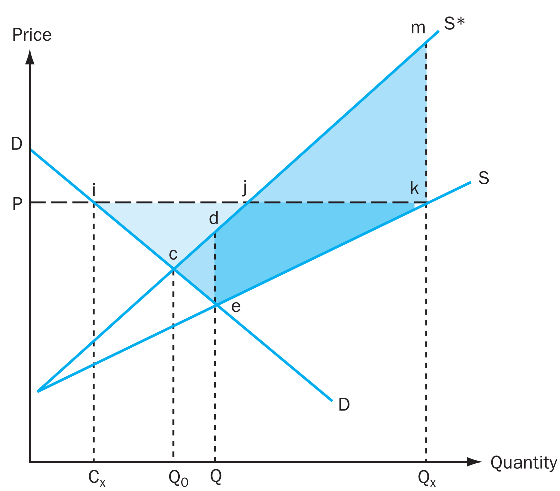


Figure 10.2 The effect of free trade in a commodity whose production is polluting: the exportables case

If the pollution tax is already in place when trade is liberalised then benefits from trade are secured irrespective of whether the final position of the country is one of exporter or importer. Suppose that we consider the case in which the good ends up being imported. Looking at Figure 10.1, the change in welfare is qcf. Now consider the case in which the commodity ends up being exported. Looking at figure 10.2, the change in welfare is cij. So even if production is increased to supply the export market the benefits from trade more than outweigh the monetised losses from increased pollution.

Is a pollution tax superior to a tax on exports or a subsidy to imports as means of controlling environmental problems? Suppose that rather than imposing an environmental tax on pollution, a tax is instead imposed on the exports of the polluting commodity. This reduces the price faced by domestic consumers of that commodity, meaning that they will now purchase too much of the good. More specifically, they value infra-marginal units of the good less than do foreign consumers. Accordingly, while trade taxes and subsidies can influence domestic pollution levels, they do so by imposing higher costs.

What happens to the small country when emissions taxes are placed upon the production of an environmentally damaging commodity in the rest of the world? Does this increase or decrease the benefits from free trade to the small open economy? Suppose that the country is already an exporter of the commodity. Anderson shows that an increase in the international price of the commodity increases pollution levels in the small economy. However, the benefits to the economy are always positive if the correct tax on pollution is already in place since the gains from trade always outweigh the losses from more pollution. If the appropriate tax is not already in place there could be either benefits or disbenefits since the quantity of pollution is already excessive. If, on the other hand, the country is an importer and remains an importer then the gains from trade are reduced.

To summarise, free trade is better than no trade provided that the requisite efficient environmental taxes, or their equivalent, are in place – provided, that is, that the externality is efficiently internalised. Taxes or subsidies on trade could alter domestic production levels and hence pollution but they are inferior to taxing environmental pollution. Countries have no incentive to depart from the economically efficient level of tax placed on the production of environmentally damaging goods.

Finally, what about the assumptions underlying the analysis? As developed countries raise their environmental standards the production of environmentally damaging goods will move to developing countries. Eventually developing countries will become exporters of environmentally damaging products. But any further increases in international standards will benefit developing countries. The ability to import or export factors of production – assumed absent in the foregoing discussion – in response to changes in environmental targets actually enhances the gains from trade. The assumption that pollution does not cross national boundaries is also important, since with transboundary pollution, increases in pollution affect all countries. Such transboundary pollution would, however, occur even in the absence of trade. There is no reason to suppose that international trade exacerbates this problem.

10.3 General equilibrium models of trade and the environment

Partial equilibrium models are used for the purposes of analysing the effects of liberalising trade in one sector of the economy, when that sector is small in relation to the rest of the economy. But there are important questions that a partial equilibrium approach cannot answer. What determines whether the economy will import or export a particular commodity (in fact we already know the answer to this question from Heckscher–Ohlin)? What happens when trade is liberalised across several sectors of the economy at once, such that the economy grows? What happens to the shadow price of pollution? What happens to economy-wide emissions? Answering these and other questions requires a general equilibrium approach in which all markets clear simultaneously.

In this section we will derive the general equilibrium level of pollution in an open economy as a function of the general equilibrium marginal abatement cost curve and marginal damage curve. Then we will go on to consider explicitly the general equilibrium effect of trade liberalisation on economic welfare. Throughout this section we draw extensively on the work of Copeland and Taylor (2003), especially chapters 2 and 4 of their book, and we employ the same notation as them.

Copeland and Taylor assume a small open economy facing prices p for two commodities x and y. The production of commodity x results in pollution and is referred to as the dirty good. The production of good y produces no pollution and is the clean good. The economy has at its disposal L units of labour and K units of capital. Emissions are denoted z and are proportionate to potential output of good x. Although it is clearly not the only way of allowing for abatement activity, emissions can be reduced by sacrificing some of the output of good x. Reducing emissions shifts the economy's production possibility frontier.

The authors define the national income function G of the economy which has as its arguments world prices, the level of both factor inputs, and permissible emissions

$$G = G(p, K, L, z) \tag{10.1}$$

More formally, G returns the maximum value for national income which can be achieved given prevailing world prices, available factor inputs and production technologies, and maximum allowable emissions. Differentiating the national income function with respect to emissions yields the general equilibrium marginal abatement cost function. It indicates by how much national income would increase if emissions were allowed to rise by one unit. Associated with the general equilibrium marginal abatement cost function is the marginal tax rate τ necessary to ensure that emissions do not exceed z.

$$\tau = G_z(p, K, L, z) \tag{10.2}$$

We turn now to the consumer side of the economy in which there are N individuals.

The indirect utility function V of each individual is given by

$$V = V(p, I, z) \tag{10.3}$$

Where $I = G/N$ is per capita income. The indirect utility function gives the maximum amount of utility as a function of prevailing prices, available income and emissions (which impact negatively on utility). This function is maximised subject to the constraint linking national income to permitted emissions. The first-order conditions include

$$V_p \frac{dp}{dz} + V_I \frac{dI}{dz} + V_z = 0 \tag{10.4}$$

Given the assumption that world prices are wholly independent of national emissions, rearranging equation 10.4 yields

$$\frac{dI}{dz} = -\frac{V_z}{V_I} \tag{10.5}$$

The RHS of equation 10.5 is recognisable as the Marginal Damage (MD) from emissions. This will be a function of prices, per capita income and emissions. For optimality we already know that aggregate marginal damage should be set equal to the tax on emissions, so

$$\tau = N.MD\left[p, \frac{G(p, K, L, z)}{N}, z\right] \tag{10.6}$$

The optimal level of emissions is therefore implicitly defined by

$$G_z(p, K, L, z) = N.MD\left[p, \frac{G(p, K, L, z)}{N}, z\right] \tag{10.7}$$

The optimal level of emissions and the emissions tax rate are jointly determined by the intersection of the 'supply' of emissions (i.e. the aggregate marginal damage curve) and the 'demand' for emissions (i.e. the marginal abatement cost curve).

This framework can be used to trace out the relationship between national income and emissions – the environmental Kuznets curve introduced in Chapter 2. But as Copeland and Taylor point out, there is no straightforward relationship between national income and emissions. Both z and G are endogenous variables and the precise relationship between national income and emissions depends on what underlies the economic growth.

If economic growth is caused by an accumulation of human capital (an increase in L) then the economy will increasingly specialise in the labour-intensive non-polluting commodity (this is the Rybczynski theorem). This will cause the demand for pollution to fall while at the same time the growth in income will also cause the supply of pollution to fall. These effects reinforce one another, resulting in a negative relationship between economic growth and emissions. If, on the other hand, the growth in income is caused by an increase in the capital stock then the demand for pollution increases. This may be more than sufficient to offset any income-induced reduction in the supply of emissions, pointing to a positive relationship between economic growth and emissions. Economic growth, first characterised by increased capital and latterly by increased human capital, generates the stylised inverse U-shaped relationship between income and emissions – the EKC. But that is only one of many possible outcomes.

Copeland and Taylor then go on to consider the effects of trade liberalisation using their general equilibrium model. What do they mean by trade liberalisation? Trade liberalisation means either reducing the level of some tariff which has been placed on some imported commodity or, alternatively, a reduction in trade frictions. In other words, in order to export a unit of a commodity an amount $(1 + \rho)$ units has to be shipped, where $\rho > 0$. This is known as the 'iceberg' model for obvious reasons. This generates a real resource cost to international trade, representing, perhaps, the transportation costs of international trade.

Trade frictions, point out Copeland and Taylor, create a difference between foreign and domestic prices but do not result in any government revenue. If the home country imports x the domestic price will be $p^d = p(1 + \rho)$ where p is the world price. If x is exported then $p^d = p/(1 + \rho)$. These frictions serve to reduce trade. Now, totally differentiating the national income equation, 10.1, gives for K and L constant

$$\mathrm{d}G = G_{p^d}\mathrm{d}p^d + G_z\mathrm{d}z \tag{10.8}$$

Using the result known as Hotelling's lemma (which in effect says that the derivative of G with respect to price is equal to x) and also the fact that the derivative of G with respect to z is equal to τ, and the definition of per capita income I, we can rewrite this as

$$\mathrm{d}I = \frac{x}{N}\mathrm{d}p^d + \frac{\tau}{N}\mathrm{d}z \tag{10.9}$$

Totally differentiating the indirect utility function and rearranging gives

$$\mathrm{d}V = V_I\left(\frac{Vp^d}{V_I}\mathrm{d}p^d + \mathrm{d}I + \frac{V_z}{V_I}\mathrm{d}z\right) \tag{10.10}$$

Inserting the expression for dI in equation 10.10 gives

$$\mathrm{d}V = V_I\left(\frac{Vp^d}{V_I}\mathrm{d}p^d + \frac{x}{N}\mathrm{d}p^d + \frac{\tau}{N}\mathrm{d}z + \frac{V_z}{V_I}\mathrm{d}z\right) \tag{10.11}$$

Roy's theorem says that the derivative of the indirect utility function V with respect to prices divided by the derivative of the same function with respect to income yields the negative of the demand equation. Invoking Roy's theorem and using the definition of MD gives

$$N\frac{\mathrm{d}V}{V_I} = -(x^c - x)\mathrm{d}p^d + (\tau - N.MD)\mathrm{d}z \tag{10.12}$$

Notice that we have used x^c to distinguish aggregate domestic consumption from x which is domestic output. It can be seen that this equation contains two terms on the RHS. The first term on the RHS is necessarily positive. This is because if the country imports x then $(x^c - x)$ is positive and trade liberalisation will reduce p^d. Conversely if the good is exported then $(x^c - x)$ is negative and the effect of trade liberalisation on p^d is positive. The second term will be zero if the tax rate on emissions has been set optimally. What this means is that trade liberalisation will necessarily increase welfare providing that the tax is set optimally.

Copeland and Taylor then demonstrate that the same analysis can also be carried out when the trade distortion is in the form of a tariff. In this case the definition of per capita income is expanded to include a term representing the revenues from placing a tax t on the import of good x

$$I = \frac{G(p(1 + t), K, L, z) + tp(x^c - x)}{N} \tag{10.13}$$

Proceeding exactly as before we get

$$N\frac{\mathrm{d}V}{V_I} = tp\frac{\mathrm{d}(x^c - x)}{\mathrm{d}t} + (\tau - N.MD)\frac{\mathrm{d}z}{\mathrm{d}t} \tag{10.14}$$

The change from an increase in the tariff t is necessarily negative if the emissions tax is set optimally. This is because $\mathrm{d}(x^c - x)/\mathrm{d}t$ is negative (i.e. imports fall if the tariff rises). Once more, we see that if the pollution tax is set optimally, trade liberalisation would be necessarily beneficial.

Copeland and Taylor enquire further under what circumstances the change in emissions will be positive or negative as a result of trade liberalisation. But irrespective of whether or not emissions increase or decrease following trade liberalisation, provided that the tax on emissions has been set optimally welfare is guaranteed to increase. And if emissions are controlled via a system of tradable permits then welfare necessarily improves following trade liberalisation because $dz = 0$. Trade liberalisation can therefore have quite different consequences depending on whether the country has a comparative advantage in the production of the clean good, and depending on whether environmental policy is optimally adjusted, and if not what policy instruments are used to control pollution.

10.3.1 Scale, composition and technique effects

Grossman and Krueger (1993) present a useful decomposition for thinking about the reasons underlying changes in emissions. Emissions are by definition equal to the overall scale of activity S, multiplied by the share of dirty goods in total output σ, multiplied by the emissions per unit of the dirty good, e.

$$z = S\sigma e \tag{10.15}$$

Taking logarithms and then totally differentiating this expression gives

$$\frac{dz}{z} = \frac{dS}{S} + \frac{d\sigma}{\sigma} + \frac{de}{e} \tag{10.16}$$

What this expression says is that the percentage change in emissions is equal to the percentage change in the scale of output plus the percentage change in the share of the dirty commodity plus the percentage change in emissions intensity of the dirty commodity.

For example, trade liberalisation boosts market access, which will generate economic growth. Accordingly, there is likely to be an increase in the scale of economic activity. Other things being equal, the scale effect will be environmentally damaging. Trade liberalisation is also likely to alter the composition of output. If a country enjoys a relative abundance of the environmental factor of production then that country will increasingly specialise in the production of environmentally intensive 'dirty' commodities. Similarly, if a country possesses a relative abundance of capital then that country will specialise in the production of capital-intensive commodities. In fact, many goods are both capital intensive and environmentally intensive. The overall effect of the composition effect on the environment is therefore ambiguous. Finally, insofar as trade liberalisation increases per capita income levels the public will call for greater environmental quality. Assuming that the government is responsive to such demands it will tighten environmental regulations. Put another way, less of the environmental resource will be allocated to production, thereby compelling producers to adopt different production techniques. This is the technique effect of trade liberalisation.

The detrimental impact of trade liberalisation on the scale effect, the ambiguous impact of trade liberalisation on the composition of output effect, and the likely beneficial impact of trade liberalisation on the technique effect mean that the overall impact of trade liberalisation is an empirical question. Box 10.1 describes one attempt to estimate the effect of trade liberalisation on emissions.

Box 10.1 The effect of trade liberalisation on emissions: an empirical test[1]

Cole and Elliott (2003) empirically analyse the effect of an increase in trade intensity (the ratio of imports plus exports to GDP) on the per capita emissions of four different pollutants: SO_2, NO_x, CO_2 and Biological Oxygen Demand (BOD). An increase in trade intensity is taken as synonymous with trade liberalisation. Using panel data regression techniques to account for country-specific fixed effects, they include per capita income in their regressions to capture both the scale and technique effect. They also include the capital–labour ratio to account for differences in the relative price of clean and dirty goods under autarky.

[1] We are grateful to Matthew Cole for supplying notes on which Box 10.1 is based.

Box 10.1 ***continued***

To determine the effect of an increase in trade intensity, trade intensity is interacted with a country's capital–labour ratio and with a country's per capita income. Note that because comparative advantage is a relative concept, countries' capital–labour ratios and per capita income levels are expressed relative to the world average. Theory suggests that the effects of an increase in trade intensity should differ for a country with a relatively high capital–labour ratio compared to a country with a relatively low capital–labour ratio, so these interacted terms should be statistically significant. An increase in trade intensity is further expected to increase pollution for countries with low per capita incomes (and correspondingly lax environmental regulations) and reduce pollution for those with high incomes (and correspondingly stringent environmental regulations).

In the following table, K denotes capital, L denotes Labour, INC denotes per capita income, and O denotes trade intensity.

Variable	SO_2 emissions per capita	No_x emissions per capita	CO_2 emissions per capita	BOD emissions per capita
K/L	2264.34***	4.54	12.15***	–17.27
$(K/L)^2$	–0.017***	0.00036**	0.000037	0.00026
INC	–77.5***	231.49***	72.27***	–651.04***
INC^2	0.00032	–0.00014	–0.00023	0.0082**
K × L × INC	0.59	–0.26*	–0.00508	–0.00053
O	2.63***	13.97	0.66*	–70.68***
O × relative K/L	–8.72***	159.53***	–4.47*	74.73***
O × relative $(K/L)^2$	1.82**	–90.37**	0.20	–20.68**
O × relative INC	2.81***	–92.29***	1.71*	–20.30***
O × relative INC^2	–0.48**	2.72	–0.24	0.86
O × relative K/L × relative INC	0.47	42.90	0.39	4.95
Time Trend	–300.92***	18801.64*	–185.30***	18933.51***
R^2 (overall)	0.13	0.58	0.33	0.13
Observations	104	104	672	672

Note: *** implies statistical significance at the 1% level of confidence; ** implies statistical significance at the 5% level of confidence; and * implies statistical significance at the 10% level of confidence.

With regards to the situation under autarky, Cole and Elliott find that increases in the capital–labour ratio increase per capita emissions of SO_2, NO_x and CO_2, although for SO_2 each additional increase in the capital–labour ratio has a diminishing impact. For BOD, they find no statistically significant relationship between emissions and the capital–labour ratio. The combined impact of the scale and technique effects appears to be negative for SO_2, positive for NO_x and CO_2, and curvilinear for BOD.

Turning attention to the variables interacted with trade intensity, it is immediately apparent that the effects of an increase in trade intensity do indeed depend on the relative value of the capital–labour ratio. Contrary to expectation, however, a high-income country (with a correspondingly high level of environmental regulations) will find that an increase in trade intensity increases SO_2 emissions. Nevertheless, a high-income country will experience falls in NO_x and BOD emissions in response to an increase in trade intensity.

Cole and Elliott's findings for NO_x and BOD are quite different from those for SO_2 and CO_2. For BOD, the trade intensity coefficient is negative and significant, while for NO_x it is positive but insignificant. In addition, the coefficients on the terms interacting trade intensity and the relative capital–labour ratio have opposite signs to those estimated for SO_2. One reason why the results for BOD may differ from those for the air pollutants is that there is no statistically significant correlation between those sectors that are water pollution intensive and those that are capital intensive. Furthermore, a large proportion of the emissions resulting in BOD come from the agricultural sector. The probable reason why the results for NO_x differ is the role of the transport sector in determining NO_x emissions.

Box 10.1 ***continued***

The following table reports the estimated scale and technique, and trade intensity elasticities. With regards to SO_2 emissions, an increase in income of 1% will generate a reduction in per capita emissions of 1.7% whereas the trade intensity elasticity is 0.3%. The results for BOD suggest that increased trade intensity will reduce per capita emissions, while for NO_x and CO_2 it is likely to increase emissions. It is worthwhile emphasising that these elasticities are calculated at mean values of the independent variables. Evaluating these elasticities for particular countries having very high or low relative capital–labour ratios or income levels will result in different elasticities.

Elasticity	SO_2	NO_x	CO_2	BOD
Scale and Technique	−1.7***	1.0***	0.46***	−0.06***
Trade	0.3***	0.03	0.049*	−0.05***

Note: These elasticities were evaluated at the means of the independent variables. *** implies statistical significance at the 1% level of confidence; ** implies statistical significance at the 5% level of confidence; and * implies statistical significance at the 10% level of confidence.

Finally, in an attempt to isolate the technique effect, Cole and Elliott also estimate the impact of changes in income and trade intensity on emissions per unit of output. Note that there is no scale effect here. Since emissions are scaled by output the authors are controlling for the scale effect. For all four pollutants, growth in per capita income appears to reduce the emissions intensity of output. And the trade intensity elasticity of emissions per unit of output is either negative or statistically insignificant.

Elasticity	SO_2	NO_x	CO_2	BOD
Technique	−0.70**	−0.88***	−0.10	−0.20
Trade	−0.36**	−0.52***	0.051	−0.27**

Note: These elasticities were evaluated at the means of the independent variables. *** implies statistical significance at the 1% level of confidence; ** implies statistical significance at the 5% level of confidence; and * implies statistical significance at the 10% level of confidence.

10.4 Do governments have an incentive to manipulate environmental standards for trade purposes?

Do governments distort their environmental policies for trade purposes and, if so, what can or should be done about it? Environmentalists often express the concern that governments have an incentive to engage in 'ecological dumping' i.e. lowering environmental standards in order to gain competitive advantage. We will examine this claim first in the context of a model of duopoly. We will argue that because international trade agreements severely restrict the use of more preferred policy instruments, policy makers might strategically manipulate environmental standards. But, critically, policy makers sometimes have incentives to adopt stronger rather than weaker environmental standards.

In the context of a two-stage game Barrett (1994) examines the incentives for governments to behave strategically while setting environmental standards. There are two countries and a firm in each country. Production generates environmental emissions which cause localised environmental damage even though such emissions may be abated. There are no transboundary pollution flows and for convenience consumers are located in a third country. In the first stage of the game governments choose permissible emissions. In the second stage of the game the firms choose their output.

The paper demonstrates that, depending on the form of competition that ensues, governments have an incentive to set an environmental policy that is either too weak or too strong. This means in effect that the marginal costs of abatement are not set equal to the marginal damage. Barrett also establishes that while governments have an incentive strategically to manipulate environmental standards, such a measure is always inferior to conventional export subsidies or taxes. Governments would always prefer using such measures if they were permitted.

Following closely the notation employed by Barrett, let x denote the output of the domestic firm and y the output of the foreign firm. The permissible emissions standard of the domestic firm is e. Revenue of the domestic firm is a function of both domestic and foreign output $R(x, y)$. Production costs of the domestic firm are a function of domestic output $C(x)$. Domestic abatement costs are a function of domestic output and permissible emissions allowed by the government $A(x, e)$. Domestic environmental damage is a function of domestic emissions $D(e)$. It can be assumed that $R_y < 0$. It can also be assumed that $C_x > 0$, $A_x > 0$ and $A_e < 0$. Finally, marginal environmental damage is increasing with permitted emissions so $D_e > 0$.

Let π denote the profits of the domestic firm. The objective of the firm will be to maximise the following with respect to x

$$\max \pi = R(x, y) - C(x) - A(x, e) \tag{10.17}$$

Given emissions standards the first-order condition for profit maximisation is given by

$$\pi_x = R_x - C_x - A_x = 0 \tag{10.18}$$

The domestic government chooses its emissions standard e in order to maximise net benefits (NB) composed of firm profits and environmental damage

$$\max NB = \pi(x, y, e) - D(e) \tag{10.19}$$

The first-order condition is

$$-A_e = D_e \tag{10.20}$$

This states that the marginal cost of abatement should be set equal to marginal damage. Barratt refers to this as the environmentally optimal emissions standard. However, if the government recognises that its choice of emissions standard e affects the level of x and y it may choose to take account of these in determining the appropriate standard. Then the following first-order condition arises

$$\pi_x \frac{\mathrm{d}x}{\mathrm{d}e} + \pi_y \frac{\mathrm{d}y}{\mathrm{d}x}\frac{\mathrm{d}x}{\mathrm{d}e} + NB_e = 0 \tag{10.21}$$

This equation contains three terms. The first term is equal to zero because we can rely on the firm to choose output x such that $\pi_x = 0$. We note that $\pi_y < 0$, $\mathrm{d}y/\mathrm{d}x < 0$ and $\mathrm{d}x/\mathrm{d}e > 0$ (see Barrett for further details). This makes the second term positive. The immediate implication is that a government behaving strategically would wish to set an emissions standard at which marginal damage costs exceed abatement costs. The intuition underlying this result is as follows. A marginal increase in permitted emissions has several effects. There is a marginal increase in output and a marginal increase in profits. But the expansion of output will cause the foreign competitor to reduce their output, thus increasing prices and profits further.

This is similar to the result of Brander and Spencer (1985) who, employing a similar setup, find it optimal for governments to subsidise their own firms. However, Barratt demonstrates that although the government is able to provide an implicit subsidy by increasing the limit on emissions, this has a cost in the form of domestic pollution problems. In this sense it differs from a pure subsidy. Consequently, a government would prefer to use a pure subsidy for the purpose of inducing a higher level of output if only such a measure were available to it.

Now, it seems reasonable to assume that the foreign government would also respond strategically. This gives the Nash response and ironically both firms earn lower profits and experience higher levels of pollution. However, neither country has the incentive to alter its behaviour. Barratt's paper further shows that both countries have an interest in cooperating and imposing stronger environmental standards than those corresponding to the environmentally optimal emissions standard.

Although there is policy competition between the two governments, this does not result in a race to the bottom since there is a cost to governments engaged in an attempt to gain an increased share of the world market in the form of increased levels of domestic pollution.

Finally, Barratt considers the case of Bertrand competition where firms compete over prices rather than output. The analytical model is quite similar but now with the result that the government should strategically reduce permissible emissions. This result seems surprising, but in the wider trade literature it is well known that if it is optimal to provide an export subsidy under Cournot competition, then it is optimal to impose an export tax under Bertrand competition. An obvious question is which concept – Cournot competition or Bertrand competition – is appropriate for any given market. Governments need to know the form of oligopolistic interaction being used in the market in which they want strategically to intervene.

How important are the assumptions of no domestic consumers and no transboundary pollution flows to this particular model? Ulph (2000) explains that relaxing these assumptions actually reinforces the conclusion that Governments have incentives to engage in ecological dumping. Since imperfect competition implies less output than socially desirable, governments would positively desire that their firms expand output. Similarly, where there are transboundary pollution problems governments have an incentive to reduce foreign output and hence foreign pollution flows.

We have addressed the question of whether governments have incentives strategically to manipulate environmental policy in the context of a duopoly, but do these arguments carry over to other kinds of markets? In fact, there is no strategic element in competitive markets since the world price cannot be altered. In the monopoly case there is no rival firm whose output can be manipulated. If the exporter is a monopolist serving mainly the foreign market there is really no need for the government to do anything except internalise the external costs.

There are, however, circumstances in which governments have incentives to manipulate the terms of trade in their favour but, where prevented from using export taxes and subsidies for this purpose, manipulate environmental standards as a second best. For example, a country that is a major importer of a polluting commodity will increase permissible emissions to encourage domestic production if unable to use an import tax to depress world prices. Likewise, a country which is a major exporter of a dirty commodity will reduce permissible emissions if it cannot use an export tax to raise world prices.

What then can be done about governments using environmental policy for trade purposes? One possibility is to allow countries to take action against those believed to be engaged in environmental dumping (see section 10.7 below). However, 'to substantiate such a claim it would be necessary at the very least to demonstrate that the standards are even lower than would be expected on the basis of such factors as the level of per capita income and the characteristics of the physical environment. Clearly that would be very difficult to do' (GATT, 1992).

Another possibility is to allow environmental policy to be set at the supranational level e.g. at the level of the EU. What form might such intervention take? As we have already argued, it is very difficult to support the harmonisation of biophysical standards and environmental taxes. Harmonisation is inappropriate because environmental preferences differ as well as differences in the ability of countries to dissipate waste, whereas minimum standards presumes that if one country increases its environmental standards to the required minimum, another country is not then tempted to reduce its own standards, standards that currently exceed the minimum. And would supranational organisations have all necessary information to calculate the appropriate environmental standard or would they prove too remote?

10.5 Environmental policy and competition between jurisdictions for mobile capital

The view that national and regional governments are likely to compete to attract new firms in part by relaxing environmental standards is pervasive. Numerous calls have been made for minimum standards to prevent the excessive degradation of environmental standards (e.g. Cumberland, 1981). But what precisely are the incentives to engage in a race to the bottom? If residents care about environmental quality as well as labour income then competitively lowering environmental standards patently imposes a cost on the community. Indeed, rather than a race to the bottom, arguably we are witnessing the perpetual tightening of environmental standards. But is this the consequence of federal governments actively restricting the ability of jurisdictions to use environmental regulation as a tool to attract new businesses?

In this section we describe the Oates and Schwab (1988) model of inter-jurisdictional competition. In a community composed of homogeneous and geographically immobile individuals, workers select a tax rate on capital and simultaneously the level of environmental quality. On one hand, attracting capital boosts wages. But, on the other hand, attracting capital via reducing taxes reduces non-labour income, while attracting capital via increasing permitted emissions reduces utility. To anticipate the conclusions, the model finds that only in very specific circumstances is there a tendency for governments to competitively lower environmental standards. And there are even circumstances in which governments might wish to competitively raise standards.

The assumptions of the model are as follows (once again we will use the same notation as in the Oates and Schwab paper). Individuals live and work in the same jurisdiction. Critically, there is no transboundary pollution. The local government sets a standard for environmental quality which determines permissible emissions. It is assumed that each firm is allocated emissions rights in direct proportion to the number of workers employed. The constant returns to scale production function is given by

$$Q = F(K, L, E) \tag{10.22}$$

where Q is output, F is the production function, K is capital, L is labour and E is emissions. The production function can be rewritten in intensive form (i.e. in per capita terms)

$$Q = Lf(k, \alpha) \tag{10.23}$$

Where k is capital but in per capita terms and $\alpha = E/L$. It is not inconsistent for α to represent both aggregate emissions and a specific emissions–labour ratio since the number of workers is fixed. The marginal products of capital, labour and emissions can be written as

$$\frac{\partial Q}{\partial K} = f_k \tag{10.24}$$

$$\frac{\partial Q}{\partial L} = f - kf_k - \alpha f_\alpha \tag{10.25}$$

$$\frac{\partial Q}{\partial E} = f_\alpha \tag{10.26}$$

It is assumed that $f_{kk} < 0$, $f_{\alpha\alpha} < 0$ and $f_{k\alpha} > 0$. Tax t is levied per unit of capital. Per capita tax revenue T is given by

$$T = tk \tag{10.27}$$

Capital receives its marginal product. Since the return to capital net of taxes matches the rate of return r available elsewhere, the local stock of capital adjusts until

$$f_k - t = r \tag{10.28}$$

Labour is perfectly immobile. It receives its marginal product. Notice carefully, however, that there is also an additional gain in output from the extra emissions associated with hiring an extra worker (see above). The gain from the extra emissions is given by αf_α. Together with the marginal product of labour this makes the wage rate equal to

$$w = f - kf_k \tag{10.29}$$

Representative utility of an individual worker can be written as

$$u = u(c, \alpha) \tag{10.30}$$

where c is consumption. The budget constraint of each individual is given by

$$c = y + w + T \tag{10.31}$$

where y is exogenous income. Inserting the expressions for w and T yields

$$c = y + (f - kf_k) + tk \tag{10.32}$$

We now maximise the utility of the representative individual with respect to c, α, k and t subject to the budget constraint and the required rate of return on capital to obtain the 'median voter' outcome. The first-order conditions for optimality include

$$u_c + \lambda_1 = 0 \tag{10.33}$$

$$u_\alpha - \lambda_1 f_\alpha + \lambda_1 k f_{k\alpha} + \lambda_2 f_{k\alpha} = 0 \tag{10.34}$$

$$-\lambda_1 f_k + \lambda_1 f_k + \lambda_1 k f_{kk} - \lambda_1 t + \lambda_2 f_{kk} = 0 \tag{10.35}$$

$$-\lambda_1 k - \lambda_2 = 0 \tag{10.36}$$

Substituting equation 10.36 into equation 10.35 demonstrates that the optimal tax rate t must be zero (since λ_1 cannot be zero). Also, substituting equation 10.36 into equation 10.34 and then dividing by equation 10.33 reveals the following:

$$-\frac{u_\alpha}{u_c} = f_\alpha \tag{10.37}$$

This equation states that the marginal rate of substitution between consumption and environmental quality as revealed by the LHS of the equation should be set equal to the marginal product of the environment.

If instead a positive value of t had already been chosen then using equation 10.35 to obtain an expression for $\lambda_1 k + \lambda_2$ and inserting this into equation 10.34 and dividing by equation 10.33 yields

$$-\frac{u_\alpha}{u_c} = f_\alpha - t\left(\frac{f_{k\alpha}}{f_{kk}}\right) \tag{10.38}$$

Given that the numerator of the bracketed expression is positive and the denominator is negative, the implication is that the social benefit of emissions reductions exceeds the social cost. In other words, when there is a need to raise tax revenues from taxes on capital the community will tighten environmental policy until willingness to pay equals the cost in terms of lost output and lost tax revenues. When does such a need arise? Oates and Schwab explain that this would happen in a situation in which head taxes are not possible. They also argue that where the government has the goal of maximising some weighted function of consumer utility and tax revenues, the government will set a positive tax rate on capital and allow the marginal benefit of improving the environment to exceed the marginal cost.

Oates and Schwab then go on to consider the situation in which the economy is populated by heterogeneous individuals: those who work and those who rely on revenue from taxing capital. Labourers would prefer an influx of capital that will raise wages whereas non-workers want higher capital tax revenues. What happens depends on which group is in the majority. If workers are in the majority then there will be a subsidy to capital. Workers gain the benefit of a larger capital stock in the form of higher wages but bear only a fraction of the cost of providing the subsidy. It can be shown that the marginal cost of improving the environment now exceeds the marginal costs.

To summarise, governments have no incentive to manipulate environmental standards in order to attract capital unless they are obliged to tax capital for revenue-raising purposes. In this case the marginal benefit of improving the environment should exceed the marginal cost. But in regions populated by heterogeneous individuals this rule can be reversed.

Other important contributions to the theory of inter-jurisdictional competition – there are many – include Markusen *et al.* (1993). In an interesting empirical analysis Fredrikson and Millimet (2002) find that decentralised governments respond to improved environmental standards in neighbouring jurisdictions by raising their own environmental standards. Reductions in the environmental standards of neighbours, by contrast, seem not to elicit any response.

10.6 Banning trade in endangered species

Measures to protect endangered species often include banning international trade in animal products e.g. skins and ivory, as well as better enforcement of anti-poaching regulations and compensation for preservation benefits enjoyed by rich countries. The 1973 Convention on International Trade in Endangered Species (CITES) banned trade in animal products listed in Appendix I to the agreement. Trade in the animal products of species listed in Appendix II to the agreement is also tightly regulated.

The African elephant was added to Appendix I in 1989 following a serious decline in elephant numbers during the 1970s and 1980s. Some sales of ivory from stockpiles have nevertheless been permitted. Since then elephant populations in Eastern and Southern Africa have been steadily increasing to the extent that South Africa is now considering a culling programme. According to the IUCN's African Elephant Status Report 2007 there are between 470 000 and 690 000 African elephants in the wild.

Although adding the African elephant to Appendix I has seemingly increased numbers, the following question arises: Does banning international trade in endangered species like the African elephant necessarily increase stocks?

In order to answer this question we will employ the model of Bulte and Van Kooten (1996). This model presupposes knowledge of optimal control theory, a mathematical technique more fully explained in Appendix 14.1. Assume that there is no poaching and that the country in which the elephants are located is too small to affect the price of ivory. The objective function of countries with elephants is to maximise

$$\int_0^\infty (H(y) + R(x) - D(x))e^{-rt} \tag{10.39}$$

subject to the constraint

$$\dot{x} = G(x) - y \tag{10.40}$$

The benefit H from harvesting refers to the proceeds from selling ivory. The harvest of elephants is y and the marginal benefit H_y is equal to the price of ivory. The revenue R from tourism increases with the stock x of elephants with $R_x > 0$ and $R_{xx} < 0$. The damage D refers to the harm elephants inflict on agriculture. Marginal damage is increasing in the stock of elephants so $D_x > 0$ and $D_{xx} > 0$. Finally, the growth function G describes the change in elephant numbers as a function of the (breeding) stock. We assume that $G_x > 0$. The current value Hamiltonian is given by

$$H(y) + R(x) - D(x) + \mu(G(x) - y) \tag{10.41}$$

The necessary conditions include

$$H_y = \mu \tag{10.42}$$

$$r\mu - \dot{\mu} = R_x - D_x + \mu G_x \tag{10.43}$$

In the long-run equilibrium the stock size is constant and the current value costate variable is unchanging, hence

$$rH_y + D_x = G_x H_y + R_x \tag{10.44}$$

and

$$G(x) = y \tag{10.45}$$

The first term in equation 10.44 describes the opportunity cost of not killing the marginal elephant, then selling the ivory and earning interest on the proceeds. The second term on the LHS is the marginal value of the damage done to agriculture. The first term on the RHS is the value of the elephant's progeny and the final term is the value of the marginal elephant to tourism. Solving this equation for x yields the long-run stock which, by virtue of the second equation, must equal the long-run harvest.

The effect of the trade ban can be appreciated by removing $H(y)$ from the objective function (the country can no longer sell ivory), giving

$$D_x = R_x \tag{10.46}$$

and

$$G(x) = y \tag{10.47}$$

Now, whether the stock increases or decreases as a result of the trade ban depends on the relative strength of the two terms rH_y and $G_x H_y$. If $r > G_x$ then the removal of these two terms means that the LHS of equation 10.44 must increase and the RHS must decrease. This requires an increase in stock size. But if $r < G_x$ then a trade ban will actually reduce stock size.

What happens if the assumption of a small country is relaxed? What if we relax the assumption that there is no poaching? Van Kooten and Bulte (2000) extend the model to incorporate these possibilities and demonstrate that the ambiguous effects of a trade ban remain. In particular, they draw attention to the changed incentives of the government to limit the activities of the poachers under a trade ban.

The implications of this admittedly simple model demonstrate there can be no presumption that banning trade will necessarily increase the stock of an endangered species. At a minimum it is necessary to know about the prevailing interest rate and the growth rate of the species before recommending a trade ban.

10.7 The General Agreement on Tariffs and Trade and the World Trade Organisation

The objective of the General Agreement on Tariffs and Trade (GATT) is the reduction of barriers to international trade achieved through the gradual elimination of tariffs, quantitative restrictions and subsidies. Although it had a secretariat based in Geneva, the GATT was a treaty and not an international body. The GATT treaty was created in 1947 and lasted until 1995 when it was subsumed into the World Trade Organisation (WTO). Discussing the lowering of tariffs is undertaken in periodic

Box 10.2 The text of GATT article XX

'Subject to the requirement that such measures are not applied in a manner which would constitute a means of arbitrary or unjustifiable discrimination between countries where the same conditions prevail, or a disguised restriction on international trade, nothing in this agreement shall be construed to prevent the adoption or enforcement by any contracting party of measures:

. . .

(b) necessary to protect human, animal or plant life or health;

. . .

(g) relating to the conservation of exhaustible natural resources if such measures are made effective in conjunction with restrictions on domestic production or consumption.'

Source: the WTO website: http://www.wto.org/

rounds of negotiations frequently extending over several years. Recent rounds have dealt with amongst other things trade in services, agricultural products and property rights. The latest of these, the Doha round, which involved discussions between 141 countries, collapsed in 2008. Amongst other things the WTO administers trade agreements, serves as a forum for negotiations and the resolution of disputes, monitors trade policies, provides assistance to Less Developed Countries and cooperates with other international bodies.

The original GATT agreement contained a list of 38 articles of which the two most important are the most favoured nation provision (Article I) and the national treatment provision (Article III). Under the GATT agreements, countries cannot discriminate between trading partners. What this means is that imports from any contracting party should be treated no less favourably than like products imported from any other contracting party. Imported and locally produced goods should be treated equally. A dispute-handling procedure is necessary when conflicts cannot be resolved. GATT Article XXIII deals with the resolution of disputes. For our purposes, however, the most interesting provision is Article XX dealing with measures to protect human, animal or plant life or health. Box 10.2 gives extracts from the text of Article XX.

10.7.1 Tuna and dolphins

A dispute between Mexico and the US over tuna and dolphins has come to assume particular importance in how many environmentalists view the GATT and the WTO.

When tuna are fished using purse seine nets dolphins can become unintentionally trapped and suffocate. The US Marine Mammal Protection Act (MMPA) sets out standards for the protection of dolphins in the Eastern Tropical Pacific Ocean. If a country exporting tuna to the US cannot prove that its standards meet those enshrined in US law then the US must embargo all imports of tuna and tuna products from that country and any intermediary countries that purchase tuna from the country subject to the embargo. The US embargoed tuna from Mexico, Venezuela and Vanuatu, as well as a number of intermediary countries, because the methods used to catch the tuna in these countries did not meet the requirements of the MMPA.

In 1991 Mexico alleged that the US embargo on their tuna exports was inconsistent with the provisions of GATT. The dispute settlement panel found that article III requiring that imported products be accorded a no less favourable treatment than domestic products had indeed been violated. The US could not curtail imports of tuna from Mexico simply because the fishing method employed did not satisfy regulators in the US. The dispute settlement panel also declined to uphold the embargo under article XX since although the article dealt with measures necessary to protect animals and exhaustible natural resources, this did not permit a country to enforce measures falling outside its territorial jurisdiction.

Another issue raised involved the US Dolphin Protection Consumer Information Act. This required that if tuna and tuna products are to be labelled dolphin friendly then they must meet certain dolphin protection standards. By contrast this practice, which applied to tuna products irrespective of their country of origin, was found to be consistent with GATT provisions. The European Union also brought a case against the US in an attempt to overturn the US embargo on tuna products from countries that trade tuna with Mexico. This case was also upheld.

The argument of the dispute panel aimed to avoid countries attempting to impose ethical standards on other countries over which they have no jurisdiction. This would have invited a flood of restrictions, many as a pretext for reducing competition from international trade. It is clear that the dispute resolution panel took a very strict interpretation of article XX(b) and XX(g) which did not explicitly rule out extra-territorialism. The dispute resolution panel hinted that had there been an international dolphin protection agreement things might have been different.

Although the US and Mexico ultimately decided to settle their dispute bilaterally, the tuna dolphin case seemed to have troubling consequences. Environmental impacts often cross national boundaries. Surely a country can have a legitimate interest in environmental events occurring in areas outside its legal jurisdiction? Furthermore, the dispute resolution panel seemed not to distinguish between environmental resources which were not in the jurisdiction of any state, such as atmospheric or oceanic resources, and those that were within the jurisdiction of a state. The GATT secretariat soon after made it explicitly clear that it was 'not possible under GATT rules to make access to one's own market dependent on the domestic environmental policies or practices of the exporting country'.

The dispute resolution panel did not have any particular scientific expertise. Were they operating outside their technical competence? Were they too narrowly focused on trade, giving too much weight to market access and not enough to the environment? Were these people democratically accountable? Was it appropriate to distinguish between product standards and regulations aimed at production processes? What about psychological harms caused by knowing that other countries are damaging the environment and mistreating animals?

10.7.2 A critique of Article XX

Numerous articles discuss Article XX, including Bhagwati and Srinivasan (1996), Charnowitz (1991) and Esty (1994).

For a measure to be justified under Article XX it must be first shown that it falls under at least one of ten exceptions under Article XX of which only (b) and (g) allude to the environment, and then that the measure satisfies the introductory paragraph (referred to as the 'chapeau' of Article XX), i.e. that it does not constitute 'a means of arbitrary or unjustifiable discrimination between countries where the same conditions prevail' and is not 'a disguised restriction on international trade'.

Note first that article XX does not even use the word 'environment'. Likewise, article XX does not make clear what it means by 'exhaustible' natural resources. Obviously, the GATT was drafted at a time when the environment was not the priority that it is today. To be justified under article XX(b) measures must furthermore be 'necessary'. This could mean 'least inconsistent with GATT policies'. But it could also mean 'based on scientific principles'.

The insistence that governments should employ measures that are 'least inconsistent with GATT policies' is hard to understand. Obviously in any given circumstance there are likely to be a range of measures capable of securing a particular outcome. But are they equivalent from a welfare point of view? This appears to force the dispute resolution panel into making normative trade-offs. Thailand famously sought a ban on the import of foreign cigarettes on grounds of public health. But while accepting the scientific evidence linking tobacco consumption with adverse impacts on human health, the dispute resolution panel found that such a measure was not necessary since other non-discriminatory methods could be used to secure public health goals (Thailand was proposing to continue with domestic production of cigarettes).

The suggestion that scientific evidence is required before a commodity can be restricted even on a non-discriminatory fashion raises further issues. Often there is a difference between the objective and subjective assessment of risk. And of course there is scope for scientists to disagree.

Environmentalists' antipathy towards the GATT and the WTO seems partly based on the belief that trade causes environmental degradation and competitive lowering of environmental standards. Trade restrictions are necessary to promote environmental protection and to serve as sanctions for environmental treaties. But elsewhere in this chapter we have tried to explain why such beliefs might be misplaced.

It is necessary to remember that the GATT does not prevent countries from using market-based instruments or traditional command and control techniques for the purposes of environmental protection. Neither does the GATT prevent a country from dealing with environmental problems arising from goods whose production is linked to environmental externalities, whether these goods are produced domestically or imported. Furthermore, it seems impossible to believe that the WTO would block the adoption of environmental policies enjoying broad multilateral support. This is just as well since many multilateral agreements have trade provisions e.g. the Montreal Protocol and CITES.

What is not acceptable under GATT rules is to restrict trade of a good made by a particular process deemed environmentally unacceptable. Neither is it permissible to restrict trade on grounds that the production process results in transboundary pollution or other cross-border spillover effects. Whereas the first of these might seem acceptable in the case of production processes which are purely localised, the second one does not. The absence of any clause in Article XX allowing trade sanctions to be used in the case of pollution spillovers or degradation of the global commons is troubling to us.

There is, nevertheless, justifiable concern about environmental issues being exploited by those with protectionist motives. We have already explained the risk that environmental regulations will be used in place of the kind of export taxes and subsidies that the GATT has done so much to combat. The WTO does not want to see its work undone by allowing countries to adopt protectionist measures introduced ostensibly in the name of environmental protection.

10.8 The empirical evidence on environmental regulations and the pattern of trade

Finally we turn to the empirical evidence. Does environmental policy influence trade flows as envisaged by the Heckscher–Ohlin theorem? Does environmental policy affect industrial location and foreign direct investment?

There is, as we have said, a widely held belief that environmental regulations have caused a loss of competitiveness. There is also the alternative view that environmental regulations have a positive impact on competitiveness. This is the so-called Porter hypothesis. Although it is always desirable to keep an open mind, it cannot be said that the Porter hypothesis enjoys widespread support among economists. Interest in the competitiveness effect of environmental regulations has grown with the reported growth in compliance costs. For a review of the literature see Jaffe *et al.* (1995).

According to Jaffe *et al.* (1995) three types of study have been attempted. Many studies attempt to explain the net exports of various commodities with, amongst other things, variables indicating the extent to which they are subject to environmental regulation. The second type of study examines changes in the world share of production of heavily regulated industries. These studies test whether the production shares of countries with relatively low environmental regulation are increasing. Other studies seek evidence that foreign direct investment is repelled by countries (or regions) characterised by onerous environmental regulations.

Many of these studies require quantitative measures of the stringency of environmental regulation. But how is such stringency to be measured? Calculating the costs of environmental regulation is easy for end-of-pipe control technologies. But it becomes much harder when environmental controls get integrated into general production technologies. It is for this reason that survey data may be inadequate. One cannot ask business representatives what proportion of their firm's costs are the result of complying with environmental regulation. And in any case, what is the baseline against which such costs are to be assessed – the complete absence of environmental regulation? And should such control costs include expenditures on health and safety in the workplace?

There are no internationally comparable data on environmental compliance control costs. Data on environmental control costs are unavailable for most Less Developed Countries. This is a significant problem for the following reason. The degree to which domestic regulatory costs will affect the pattern of trade depends on the regulatory costs that foreign countries impose on their industries. These problems aside, what emerges is the fact that environmental control costs are generally very small indeed, apart from for a handful of industries (chemicals, petroleum, pulp and paper, and primary metals).

Turning now to the empirical evidence itself, a study by Kalt (1988) found that changes in net exports of 78 industrial commodities do not correspond to changes in environmental compliance costs over the period 1967–1977. Tobey (1990) finds that environmental regulation did not influence trade flows for 24 pollution-intensive industries and 23 countries. Grossman and Krueger (1993) find that pollution control costs do not explain the pattern of trade between the US and Mexico. This is an important study because (a) these countries are part of the NAFTA, and (b) the difference in environmental regulations between the US and Mexico are likely to be large and obviously these countries are, by virtue of their size and proximity, major trading partners. Evidence of the effect of environmental regulations on competitiveness should be visible on the US–Mexico border if it is to be seen anywhere. Van Beers and van den Bergh (1997) focus on bilateral trade flows. Unusually they find significant effects for environmental policy on trade flows, although the measure of the stringency of environmental regulation that they use is essentially arbitrary.

Ederington and Minier (2003) argue that the stringency of environmental regulations is likely to be endogenous. Given the model of strategic environmental policy presented in section 10.4, this should come as no surprise to the reader. The endogeneity of environmental regulation may have biasd downward previous estimates of the effect of environmental regulation, particularly if governments relax the environmental regulations facing industries confronting strong import competition. The authors demonstrate that accounting for policy endogeneity makes a substantial impact on the quantitative significance of environmental regulations in determining trade patterns (see also Levinson and Taylor, 2008).

Recently Ederington *et al.* (2005) have identified more precisely the conditions under which environmental regulations are likely to affect the pattern of trade. First of all, most trade takes place between developed countries which are, incidentally, also likely to have similar standards of environmental regulation. Accordingly, the impact of environmental regulations only becomes apparent when one examines trade between countries at different stages of development. There is also the case of industries protected by high transport costs, fixed plant costs and enjoying cost advantages from agglomeration. Such industries will be necessarily less sensitive to differences in environmental regulations than their more footloose counterparts. Finally, there is the fact that for the majority of manufacturing environmental costs are only a very small fraction of overall production costs. Empirical evidence suggests that distinguishing between developed and less developed countries while simultaneously accounting for differences in transport costs results in a somewhat greater role for environmental regulations.

Because new regulations might not cause existing plant to move, several authors have suggested that FDI flows are likely to be more sensitive than trade to environmental regulations. FDI decisions are of course likely to be affected by a whole range of considerations, including proximity to markets, the availability of supplies and natural resources, transport and telecommunications infrastructure, and the availability of a well-educated workforce. Isolating the effect of environmental regulations against this backdrop poses a considerable challenge, not least because environmental regulations are, as we have noted, only a small fraction of production costs.

Kolstad and Xing (2002) bring to light some evidence that FDI is attracted by the laxity of environmental regulations, at least for the two industries exhibiting the highest control costs in the US (chemicals and primary metals). Surveys of business people contemplating investment overseas, however, uniformly suggest that environmental regulations are unimportant. And any multinationals attempting to exploit lower environmental standards in developing countries might nowadays find themselves subject to considerable adverse publicity. Other studies consider the location of new plants within a country e.g. within the US. Levinson (1996) finds that environmental stringency has a negative impact on the opening of new plants across the US states. In their study of FDI in Côte d'Ivoire, Morocco, Venezuela and Mexico, Eskeland and Harrison (2003) find scant evidence that foreign investors are more likely to invest in sectors characterised by low abatement costs. Indeed, foreign-owned plants are much cleaner than their local peers.

Overall there is little evidence that trade patterns and FDI flows respond to differences in environmental regulation. But at the same time there is no satisfactory measure of the stringency of environmental regulation. The difficulty of conducting econometric studies has encouraged researchers to resort to simulation models in which the consequences of changes in the level of environmental resource endowments can be seen much more clearly.

10.8.1 CGE models

CGE models were introduced in Chapter 8 and employed in Chapter 9 to investigate the costs of economy-wide abatement of GHG emissions. Another major application of CGE models in the domain of environmental economics has been to examine the consequences for international trade of unilateral or coalition-wide attempts to reduce carbon emissions while the remainder of the world is not subject to any carbon emissions limits.

We already noted that trading in carbon emissions permits equalises the marginal costs of abatement and minimises compliance costs. The issue we wish to discuss here is the extent to which unilateral measures, or measures taken by a combination of countries, to reduce the emission of carbon might be offset by increases in emissions from other countries. This could happen either through the fall in world energy prices or the relocation of energy-intensive industry and subsequent importation of energy-intensive commodities. This phenomenon has been dubbed 'carbon leakage' and is of course a manifestation of the Heckscher–Ohlin trade theorem. Manne (1994) finds that a 20% cut in emissions from the OECD generates a leakage rate of 30%. In other words, the effective cut in emissions from the OECD is only 14%.

The assumption that a foreign-produced commodity is a perfect substitute for the same commodity produced domestically is inconsistent with the phenomenon of intra-industry trade, where the same goods and services are simultaneously imported and exported. Accordingly, modern CGE models assume that domestic and foreign goods are imperfect substitutes (this is known as the Armington assumption). The response to changes in climate policy is more marked the higher the degree of substitution between foreign and domestic carbon-intensive goods. The degree of carbon leakage will also depend on the countries in which carbon abatement takes place and whether these countries have trading partners providing goods that are close substitutes for domestically produced carbon-intensive commodities.

Bergman (1991) conducts an experiment using a CGE model for Sweden in which SO_x, NO_x and CO_2 emissions are cut by 44, 57 and 28% respectively. Although the aggregate impact of these measures is small, one of the sectors, 'steel and chemicals', has its output cut by 57%. Paradoxically, the cost of unilateral attempts to reduce carbon emissions is buffered by international trade with partners who are themselves not subject to emissions limits. For example, Burniaux *et al.* (1992) found that stabilising emissions from the OECD caused a reduction in income of 0.6%. However, for global stabilisation the cost to the OECD was increased to 1.2%.

Attempts to reduce carbon emissions will also affect the terms of trade. A worldwide attempt to reduce carbon emissions might substantially reduce the producer price of carbon energy products on world markets. This would be advantageous to those countries who, like Japan, have few indigenous carbon energy resources and import most of their oil. The OPEC countries and Australia, as a major coal producer, would lose heavily and this might explain their well-known attitude to such measures.

We reiterate the caveats already expressed in Chapter 8 concerning CGE modelling. CGE models may omit important features of the way which economies function, and depend on maintained assumptions regarding the structure and parameterisation of such models. The seemingly clear-cut results from CGE modelling exercises risk providing policy makers with a misleading sense of accuracy and confidence. But even if we are unsure whether or not we believe the results of CGE modelling exercises, they nonetheless illustrate well a range of theoretical propositions relating to trade and the environment.

Summary

Basic trade theory suggests that countries will specialise in those commodities that are intensive in their most abundant factor of production. Countries can be considered abundant in environmental resources if they are sparsely populated, possess environments capable of dissipating environmental residues and have populations that do not much care for environmental quality, perhaps because of low incomes. Such inter-country differences as exist make it hard to sustain calls for the harmonisation of environmental standards.

A partial equilibrium model suggests that trade liberalisation could have a positive or negative effect on the environment, but that even if it did have a negative effect this might well be outweighed by the real resource gains. And even this could be avoided by imposing the appropriate environmental tax from the outset. In a general equilibrium model, changes in trade frictions or resource endowments cause changes in environmental emissions. These can be usefully decomposed into changes arising from the scale of economic activity, changes in the composition of output and changes in production technique.

It is possible to construct models in which there are incentives to lower environmental standards for trade purposes but this generally occurs because countries are not allowed to use more conventional measures. Depending on the precise form that competition takes, a country might even have incentives to raise environmental standards. Likewise there are circumstances in which governments might wish to lower environmental standards in order to attract capital inflows. This could occur if the government is obliged to tax capital. But there is an optimal degree of relaxation involved and certainly no 'race to the bottom'.

Banning trade in animal products may raise or lower the stock of the species depending amongst other things on the prevailing interest rate and the species' growth rate. The GATT is required to deal with instances in which calls for environmental protection have protectionist motives. There are nevertheless legitimate questions about whether Article XX in the 1947 GATT treaty is adequate to deal with transboundary and cross-border environmental problems.

Empirical studies have generally found little evidence to suggest that patterns of trade are determined by environmental standards in the manner suggested by theory. But the data used to measure the stringency of environmental regulations are not very good and recent analyses accounting for the endogeneity of environmental policy and the differential impact of environmental regulations across industries have met with more success.

Although it might be correct in particular instances, the belief that trade is bad for the environment, that countries are forced to lower environmental standards, that the WTO undermines attempts to improve environmental quality and that footloose firms exploit differences in environmental regulations is not generally correct.

Further reading

Numerous textbooks are available for students wishing to study international trade theory. We like Bhagwati and Srinivasan (1983), but any of these will explain the classical theorems such as Heckscher–Ohlin. The classic partial equilibrium analysis of environmental externalities and the welfare effects of trade liberalisation remains Anderson (1992). Currently the best book about the trade and the environment from a general equilibrium perspective is without doubt Copeland and Taylor

(2003), although this is best suited to graduate students. The chief reference on strategic environmental policy that we used is Barrett (1994). Ulph (2000) provides an overview of the literature. For inter-jurisdictional competition the seminal paper is Oates and Schwab (1988). Turning now to CITES and the effect of trade bans on animal products, students are advised to visit the CITES website. This describes the evolution of the convention and lists all species included in the various annexes to the 1973 agreement. Van Kooten and Bulte (2000) is an interesting book on the economics of managing biological assets, with a section on the effect of trade bans. There are many documents on the WTO website explaining the organisation's activities. Also available online is the original 1947 GATT document. This document includes the infamous Article XX in its entirety. The website explains the WTO's latest thoughts on trade and the environment and the outcome of recent disputes. See also Esly (1994). There are numerous reviews of the empirical evidence testing the hypothesis that differences in the stringency of environmental regulations provide a basis for trade and FDI flows. The best of these is Jaffe *et al.* (1995).

Discussion questions

1. 'I think the economic logic behind dumping a load of toxic waste in the lowest wage country is impeccable and we should face up to that.' Do you think that the economic reasoning displayed here is right or wrong? This is another quotation taken from an infamous memo allegedly written by Lawrence Summers. The full memo can be found on our website (as well as many other internet sites).
2. Does international trade compel countries to lower environmental standards?
3. Do you think that it is acceptable for the EU to try to ban US beef containing growth hormones? Do you think that such a manoeuvre would be acceptable under GATT rules?
4. How would you test empirically the hypothesis that differences in environmental regulations provide a basis for trade? What difficulties would you expect to encounter?

PART III Project appraisal

CHAPTER 11 Cost–benefit analysis

Almost all economists are intellectually committed to the idea that the things people want can be valued in dollars and cents. If this is true, and things such as clean air, stable sea levels, tropical forests and species diversity can be valued that way, then environmental issues submit – or so it is argued – quite readily to the disciplines of economic analysis . . . most environmentalists not only disagree with this idea, they find it morally deplorable.

***The Economist*, 31 January 2002**

Learning objectives

In this chapter you will

- learn about the conditions necessary for intertemporal efficiency
- revisit the analysis of optimal growth introduced in Chapter 3
- find out how to do project appraisal
- learn about cost–benefit analysis and its application to the environment
- be introduced to some alternatives to cost–benefit analysis

Introduction

By 'cost–benefit analysis' we mean the social appraisal of investment projects. Here, 'social' signifies that the appraisal is being conducted according to criteria derived from welfare economics, rather than according to commercial criteria. Cost–benefit analysis, that is, attempts to appraise investment projects in ways that correct for market failure. If there were no market failure, social and commercial criteria would coincide. An 'investment project' is something that involves a current commitment with consequences stretching over future time. It need not, that is, be an 'investment' in the sense of the accumulation of capital, though, of course, it may be and frequently is. As generally used, the term 'cost–benefit analysis' would also embrace, for example, the appraisal of the adoption now of a government policy intended to have future effects. Also, as generally used, the term refers to the analysis of projects that are marginal with respect to the economy as a whole. A policy decision intended to change the nature of the economy, such as abandoning the market system in favour of command and control, is not marginal. A policy decision to introduce a new form of taxation would be marginal. An investment project, such as a new nuclear power plant or a new airport, could be large in absolute terms, but would nonetheless be a small part of total investment, and hence marginal.

Cost–benefit analysis relates to the environment in two main ways. First, many projects intended to yield benefits in the form of the provision of goods and services have environmental impacts – consider damming a river in a wilderness area to generate electricity. To the extent that such impacts are externalities (see Chapter 4) there is market failure and they do not show up in private, commercial, appraisals. The costs of such projects are understated in ordinary financial appraisals. Second, there are projects the main purpose of which is to have beneficial environmental impacts – consider the construction of a sewage treatment plant. Here also the impacts typically involve external effects, and so would not appear in an ordinary financial appraisal. Projects of the first sort, where the environmental market failure involves incidental damage, may arise in both the private and public sectors. In the dam case, for example, there is saleable output and the project could be privately or publicly financed. Projects of the second type, those intended to provide environmental benefits, typically come up as public-sector projects – they provide outputs which are (again see Chapter 4) public goods. There are, of course, projects which have both desirable and undesirable impacts on the environment – waste incinerators are intended to reduce the need for landfill disposal but they generate atmospheric emissions.

In all cases, the basic strategy of cost–benefit analysis is the same. It is to attach monetary values to the environmental impacts, desired and undesired, so that they are considered along with, and in the same way as, the ordinary inputs (labour, capital, raw materials) to and outputs (goods and/or services) from the project. In this chapter we are primarily concerned with the rationale for, and the methods of, cost–benefit analysis in relation to the environment. The methods which economists have developed to value the environment so that it can be accounted for in cost–benefit analysis are dealt with principally in the next chapter, Chapter 12, but also come up in Chapter 13.

This chapter is organised as follows. As noted above, cost–benefit analysis is based on welfare economics. Also as noted above, it is essentially about dealing with situations where the consequences of a decision are spread out over time. Our previous treatment of welfare economics, in Chapter 4, ignored the temporal dimension. Hence, the first thing to be done here is to review the basics of intertemporal welfare economics. The second section of the chapter builds on that review to discuss the economics of project appraisal, starting with the private and moving from there to social appraisal, i.e. cost–benefit analysis. The third section then looks specifically at cost–benefit analysis and the environment, and considers some of the objections that have been raised about the basic idea of dealing with environmental impacts in the same way as 'ordinary' commodities. It also looks briefly at some alternative models for social decision-making where environmental impacts are important.

Finally here a word about terminology. What we call 'cost–benefit analysis' some writers refer to as 'benefit–cost analysis' – CBA, as we shall henceforward refer to it, is the same thing as BCA. Cost-effectiveness analysis is not the same thing as CBA, and we will discuss it briefly towards the end of this chapter.

11.1 Intertemporal welfare economics

Chapter 4 introduced the basic ideas in welfare economics in a timeless context. Those basic ideas, such as efficiency and optimality, carry over into the analysis of situations where time is an essential feature of the problem. In Chapter 4, we saw that efficiency and optimality at a point in time require equality conditions as between various rates of substitution and transformation. Once the passage of time is introduced into the picture, the number and range of such conditions increases, but the intuition as to the need for them remains the same. In going from intratemporal, or static, to intertemporal, or dynamic, welfare economics we introduce some new constructions and some new terminology, but no fundamentally new ideas.

The primary motivation for this discussion here of intertemporal welfare economics is to provide the foundations for an appreciation of CBA. It should be noted, however, that intertemporal welfare economics is also the background to much of the analysis of natural resource exploitation economics to be considered in Part IV of this book. We also drew upon some of the material to be presented here in our discussion of some aspects of the ethical basis for the economic approach to environmental problems in Chapter 3. Appendices 11.1 and 11.2 work through the material to be discussed in this section using the Lagrangian multipliers method in the same way as was done in the appendices to Chapter 4. The reader might find it helpful at this point to quickly revisit Chapter 4 on efficiency and optimality, and the way in which, given ideal circumstances, a system of markets could produce an efficient allocation.

11.1.1 Intertemporal efficiency conditions

In Chapter 4 we considered a model economy in which two individuals each consumed two commodities, with each commodity being produced by two firms using two scarce inputs. Appendices 11.1 and 11.2 consider that model generalised so that it deals with two periods of time. Also considered there are some specialisations of that model, which bring out the essentials of intertemporal allocation issues while minimising the number of variables and notation to keep track of. In the text here we will just look at a special model so as to deal with the essentials in the simplest possible way. Readers are, however, advised to work through the more general treatment in the appendices so as to appreciate the ways in which what follows is special.

We consider two individuals and two time periods, which can be thought of as 'now' and 'the future' and are identified as periods 0 and 1. Each individual has a utility function, the arguments of which are the levels of consumption in each period:

$$U^A = U^A(C_0^A, C_1^A)$$
$$U^B = U^B(C_0^B, C_1^B) \tag{11.1}$$

As in Chapter 4, an allocation is efficient if it is impossible to make one individual better off with-out thereby making the other individual worse off. Here, the allocation question is about how total consumption is divided between the two individuals in each period, and about the total consumption levels in each period, which are connected via capital accumulation. In order to focus on the essentially intertemporal dimensions of the problem, we are assuming that there is a single 'commodity' produced using inputs of labour and capital. The output of this commodity in a given period can either be consumed or added to the stock of capital to be used in production in the future. We shall assume that the commodity is produced by a large number of firms.

Given this, efficiency requires the satisfaction of three conditions:

1. equality of individuals' consumption discount rates;
2. equality of rates of return to investment across firms;
3. equality of the common consumption discount rate with the common rate of return.

We will now work through the intuition of each of these conditions. Formal derivations of the conditions are provided in Appendix 11.1.

11.1.1.1 Discount rate equality

This condition concerns preferences over consumption at different points in time. Figure 11.1 shows intertemporal consumption indifference curves for A and B. The curve shown in panel a, for example, shows those combinations of C_0^A and C_1^A that produce

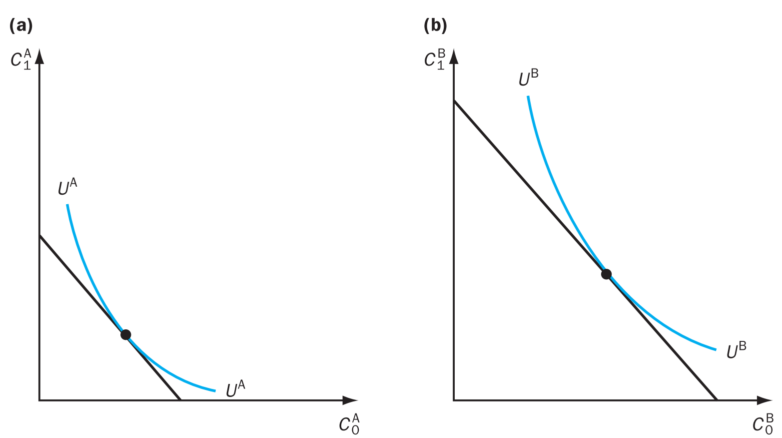


Figure 11.1 Equality of consumption discount rates

a constant level of utility for A. The curve in panel b does the same thing for individual B. In Chapter 4 we worked with marginal rates of utility substitution which are the slopes of indifference curves, multiplied by –1 to make them positive numbers. We can do that here, defining $\text{MRUS}^{\text{A}}_{C0,C1}$ in terms of the slope of a panel a indifference curve and $\text{MRUS}^{\text{B}}_{C0,C1}$ in terms of the slope of a panel b indifference curve. Given that, we can say that for an allocation to be intertemporally efficient it is necessary that

$$\text{MRUS}^{\text{A}}_{C0,C1} = \text{MRUS}^{\text{B}}_{C0,C1}$$

where the intuition is the same as in the static case – if the marginal rates of utility substitution differ, then there exists a rearrangement that would make one individual better off without making the other worse off. In fact, Figure 4.1 applies here if we just treat X there as period 0 consumption and Y there as period 1 consumption.

Following the practice in the literature, we state this condition using the terminology of consumption discount rates. For example, A's consumption discount rate is defined as

$$r^{\text{A}}_{C0,C1} \equiv \text{MRUS}^{\text{A}}_{C0,C1} - 1$$

i.e. the consumption indifference curve slope (times –1) minus 1. In that terminology, and dropping the subscripts, the intertemporal consumption efficiency condition is:

$$r^{\text{A}} = r^{\text{B}} = r \tag{11.2}$$

Note that although consumption discount rates are often written like this, they are not constants – as Figure 11.1 makes clear, for a given utility function, the consumption discount rate will vary with the levels of consumption in each period.

The reader will recall that we discussed discount rates in Chapter 3. It is important to be clear that the discounting we have just been discussing here is different from that discussed in Chapter 3. Here we have been discussing consumption discounting, there we discussed utility discounting. Different symbols are used – ρ for the utility discount rate, r for the consumption discount rate. You might expect, given that utility is related to consumption, that ρ and r are related. They are. We discuss the relationship between the utility and consumption discount rates in section 11.1.4.2 below.

11.1.1.2 Rate of return equality

This condition concerns the opportunities for shifting consumption over time. Consider the production of the consumption commodity in periods 0 and 1 by one firm. At the start of period 0 it has a given amount of capital, and we assume that it efficiently uses it together with other inputs to produce some level of output, denoted Q_0. That output can be used for consumption in period 0 or saved and invested so as to increase the size of the capital stock at the start of period 1. In Figure 11.2 $\bar{C}_0$ is period 0 consumption output from this firm when it does no investment. In that case, the capital stock at the start of period 1 is the same as at the start of period 0, and $\bar{C}_1$ is the maximum amount of consumption output possible by this firm in period 1. Suppose that all of period 0 output were invested. In that case the larger capital stock at the start of period 1 would mean that the maximum amount of consumption output possible by this firm in period 1 was $C_1^{\max}$. The solid line $C_1^{\max}A$ shows the possible combinations of consumption output in each period available as the level of investment varies. It is the consumption transformation frontier.

Figure 11.2 shows two intermediate – between zero and all output – levels of investment, corresponding to C_0^{a} and C_0^{b}. The levels of investment are, respectively, given by the distances $C_0^{\text{a}}\bar{C}_0$ and $C_0^{\text{b}}\bar{C}_0$. Corresponding to these investment levels are the period 1 consumption output levels C_1^{a} and C_1^{b}. The sacrifice of an amount of consumption $C_0^{\text{b}}C_0^{\text{a}}$ in period 0 makes available an amount of consumption $C_1^{\text{a}}C_1^{\text{b}}$ in period 1. The rate of return to, or on, investment is a proportional measure of the period 1 consumption payoff to a marginal increase in period 0 investment. It is defined as

$$\delta \equiv \frac{\Delta C_1 - \Delta I_0}{\Delta I_0}$$

where ΔC_1 is the small period 1 increase in consumption – $C_1^{\text{a}}C_1^{\text{b}}$ for example – resulting from the small period 0 increase ΔI_0 in investment which corresponds to $C_0^{\text{b}}C_0^{\text{a}}$. The increase in investment ΔI_0 entails a change in period 0 consumption of equal size and opposite sign, i.e. a decrease in C_0. With ΔI equal to ΔC_0, the definition of the rate of return can be written as

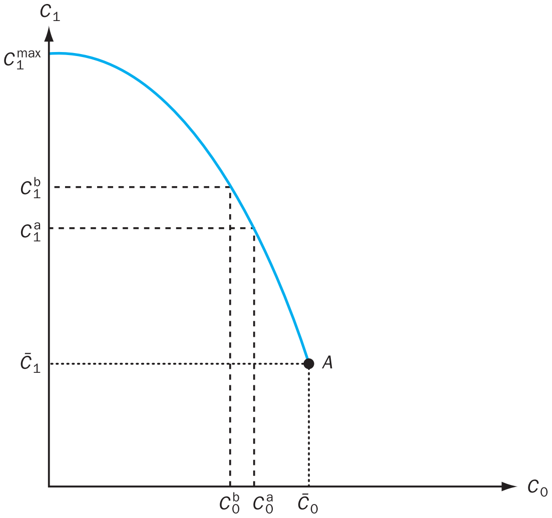


Figure 11.2 Shifting consumption over time

$$\delta = \frac{\Delta C_1 - (-\Delta C_0)}{-\Delta C_0} = \frac{\Delta C_1 + \Delta C_0}{-\Delta C_0} = -\frac{\Delta C_1}{\Delta C_0} - 1$$

which is the negative of the slope of the consumption transformation frontier minus 1. This can be written

$$1 + \delta = -s$$

where s is the slope of $C_1^{max}A$. The curvature of the line $C_1^{max}A$ in Figure 11.2 reflects the standard assumption that the rate of return declines as the level of investment increases.

Now, there are many firms producing the consumption commodity. Figure 11.3 refers to just two of them, identified arbitrarily as 1 and 2 with superscripts, and shows why the second condition for intertemporal efficiency is that rates of return to investment must be equal, as they are for C_0^{1a} and C_0^{2a}. Suppose that they were not, with each firm investing as indicated by C_0^{1b} and C_0^{2b}. In such a situation, period 1 consumption could be increased without any loss of period 0 consumption by having firm 1, where the rate of return is higher, do a little more investment, and firm 2, where the rate of return is lower, do an equal amount less. Clearly, so long as the two rates of return differ, there will be scope for this kind of costless increase in C_1. Equally clearly, if such a possibility exists, the allocation cannot be efficient as, say, A's period 1 consumption could be increased without any reduction in her period 0 consumption or in B's consumption in either period. Hence, generalising to $i = 1, \ldots, N$ firms, we have

$$\delta_i = \delta, \; i = 1, \ldots, N \tag{11.3}$$

as the second intertemporal efficiency condition.

11.1.1.3 Equality of discount rate with rate of return

If we take it that the conditions which are equations 11.2 and 11.3 are satisfied, we can discuss the third condition in terms of one representative individual and one representative firm. Figure 11.4 shows the situation for these representatives. Clearly, the point a corresponds to intertemporal efficiency, whereas points b and c do not. From either b or c it is

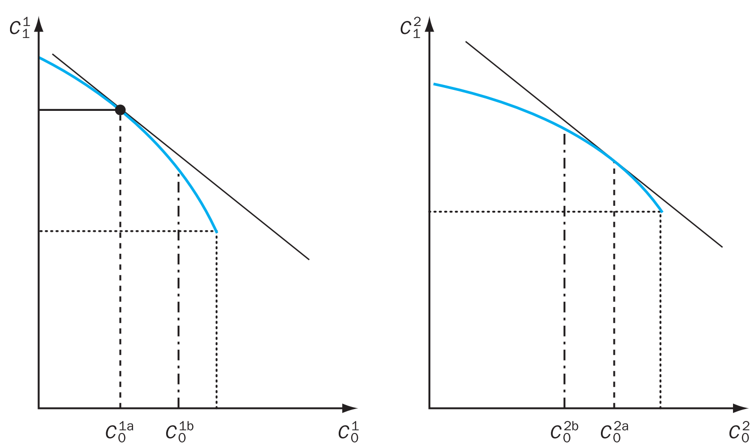


Figure 11.3 Equality of rates of return

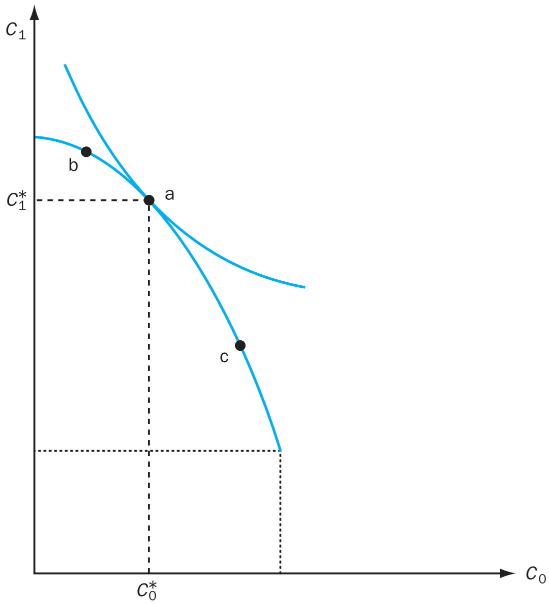


Figure 11.4 Equality of rate of return and discount rate

possible to reallocate consumption as between periods 0 and 1 so as to move onto a higher consumption indifference curve. It is impossible to do this only where, as at a, there is a point of tangency between a consumption indifference curve and the consumption transformation frontier.

At a the slopes of the consumption indifference curve and the consumption transformation frontier are equal. We have already noted that r is the slope of the former minus 1. The slope of the latter is $\Delta C_1/\Delta C_0$, so that from the definition of δ it is equal to that slope minus 1. It follows that slope equality can also be expressed as the equality of the rate of return and the discount rate:

$$\delta = r \tag{11.4}$$

11.1.2 Intertemporal optimality

In our discussion of the static, intratemporal, allocation problem in Chapter 4 we noted that efficiency requirements do not fix a unique allocation. To do that we need a social welfare function with individuals' utility levels as arguments. The situation is exactly the same when we look at intertemporal allocations. The conditions for static efficiency plus the conditions stated above as equations 11.2, 11.3 and 11.4 do not fix a unique allocation. For any given data for the economic problem – resource endowments, production functions, preferences and the like – there are many intertemporally efficient allocations. Choosing among the set of intertemporally efficient allocations requires a social welfare function of some kind.

In general terms there is nothing more to be said here beyond what was said in the discussion of the static case in Chapter 4. We shall come back to the relationship between intertemporal efficiency and optimality shortly when we make some observations on intertemporal modelling. Before that we discuss the role of markets in the realisation of intertemporal efficiency. This way of proceeding makes sense given that there is another important carry-over from the static to the dynamic analysis – while in both cases it can be claimed that market forces alone would, given ideal circumstances, realise efficiency in allocation, in neither case can it be claimed, under any circumstances, that market forces alone will necessarily bring about welfare-maximising outcomes.

11.1.3 Markets and intertemporal efficiency

Economists have considered two sorts of market institution by means of which the conditions required for intertemporal efficiency might be realised, and we will briefly look at both here. In doing that we will take it that in regard to intratemporal allocation the ideal circumstances discussed in Chapter 4 are operative so that the static efficiency conditions are satisfied. This assumption is not made as an approximation to reality – we have already seen that static market failure is quite pervasive. It is made in order to simplify the analysis, to enable us, as we did above, to focus on those things that are the essential features of the intertemporal allocation problem.

11.1.3.1 Futures markets

One way of looking at the problem of allocative efficiency where time is involved, considered in Appendix 11.1, is simply to stretch the static problem over successive periods of time. Thus, for example, we could take the economy considered in Chapter 4 – with two individuals, two commodities, and two firms producing each commodity, each using two inputs – and look at it for two periods of time. This approach could be, and in the literature has been, extended to many individuals, many commodities, many firms, many inputs, and many time periods. In following it, one thinks of the same physical thing at different times as different things. Thus, for example, the commodity X at time t is defined as a different commodity from X at time $t + 1$. This approach leads to more general versions of the intertemporal conditions stated in the previous section.

In terms of markets, the parallel analytical device is to imagine that date-differentiated things have date-differentiated markets. Thus, for example, there is market for commodity X at time t and a separate market for commodity X at time $t + 1$. It is assumed that at the beginning of time binding contracts are made for all future exchanges – the markets in which such contracts are made are 'futures markets'. Now, by this device, time has essentially been removed from the analysis. Instead of thinking about N commodities and M periods of time, one is thinking about $M \times N$ different commodities. Trade in all of these commodities takes place at one point in time. Clearly, the effect of this device is, formally, to make the intertemporal allocation problem just like the static problem, and everything said about the latter applies to the former. This includes what can be said about markets. If all of the ideal circumstances set out in Chapter 4 apply to all futures markets, then it can be formally shown that the conditions for intertemporal efficiency will be satisfied.

This is an interesting analytical construct. It will be immediately apparent that the connection between a complete set of futures markets characterised by the ideal circumstances and 'the real world' is remote in the extreme. Recall, for example, that in the static case we saw in Chapter 4 that for a pure market system to produce an efficient allocation it was necessary that all agents had complete information. In the context of the futures market construct, this involves agents now having complete information about circumstances operative in the distant future.

While futures markets do exist for some commodities – mainly standardised raw material inputs to production and financial instruments – there is very far from the complete set of them that would be required for there to be even a minimal case for seriously considering them as a means for the attainment of intertemporal efficiency. In actual market systems the principal way in which allocation over time is decided is via markets for loanable funds, to which we now turn.

11.1.3.2 Loanable funds market

We will assume, in order to bring out the essentials as simply as possible, that there is just one market for loanable funds – the bond market. A bond is a financial instrument by means of which borrowing and lending are effected. In our two-period context we will assume that trade in bonds takes place at the beginning of period 0. All bond certificates say that on day 1 of period 1 the owner will be paid an amount of money x by the bond issuer. There are many sellers and buyers of such bonds. If the market price of such bonds is established as P_B, which will be less than x, then the interest rate is:

$$i = \frac{x - P_B}{P_B}$$

A seller of a bond is borrowing to finance period 0 consumption: repayment is made on the first day of period 1, and will reduce period 1 consumption below what it would otherwise be. A buyer is lending during period 0, and as a result will be able to consume more in period 1 by virtue of the interest earned.

Now consider an individual at the start of period 0, with given receipts M_0 and M_1 at the beginning of each period, and with preferences over consumption in each period given by $U = U(C_0, C_1)$. The individual maximises utility subject to the budget constraint given by M_0 and M_1 and the market rate of interest, at which she can borrow/lend by trading in the market for bonds. Note that the individual takes the market rate of interest as given – in this context i is a constant. The individual's maximisation is illustrated in Figure 11.5. UU is a consumption indifference curve, with slope $-(1 + r)$, where r is the consumption discount rate. The budget constraint is $C_1^{max}C_0^{max}$ which has the slope $-(1 + i)$, because by means of bond market transactions $(1 + i)$ is the rate at which the individual can shift consumption between the two periods. The individual's optimum consumption levels are C_0^* and C_1^* given by the tangency of the

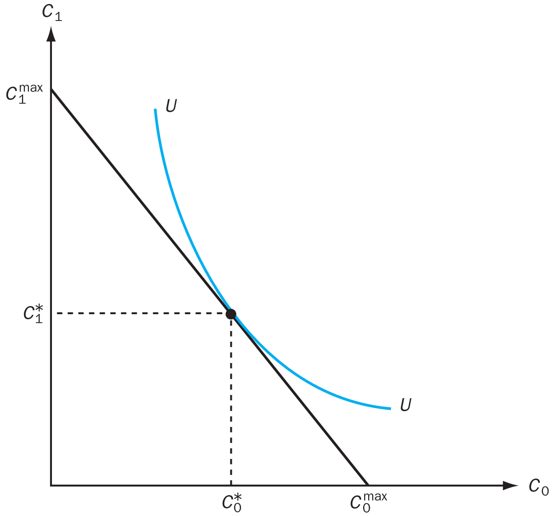


Figure 11.5 Intertemporal optimum for an individual

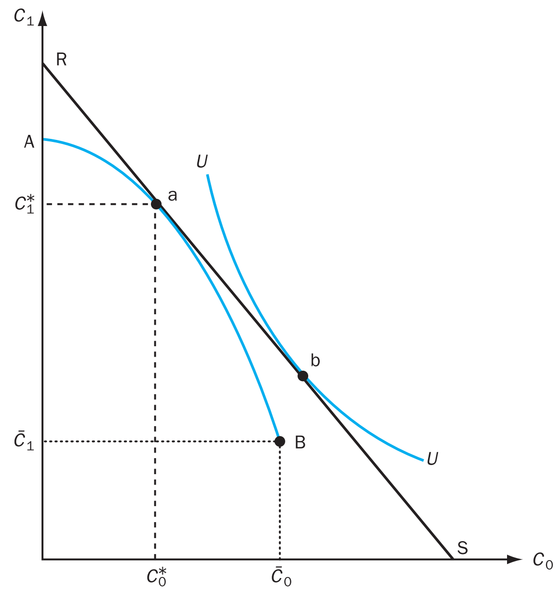


Figure 11.6 Present value maximisation

budget constraint to the consumption indifference curve. It follows that the optimum is characterised by the equality of r and i. But this will be true for all individuals, so with a single bond market clearing interest rate of i, consumption discount rates, r, will be equalised across individuals, thus satisfying the first condition for an intertemporally efficient allocation, equation 11.2. Individuals for whom C_0^* is less than M_0 will be lenders, and hence buyers in the bond market; individuals for whom C_0^* is greater than M_0 will be borrowers, and hence sellers in the bond market.

Now consider the period 0 investment decisions made by firms. The owners of firms can shift their consumption over time in two ways. First, by investing in their firm, and second by borrowing/lending via the bond market. The terms on which they can do the latter have just been discussed. What they want to do is to invest in their firm up to the point that puts them in the best position in relation to the opportunities offered by the bond market. In Figure 11.6 the curve AB shows the combinations of C_0 and C_1 available to the firm's owners as they vary their period 0 investment in the firm from zero, at B, to the maximum possible, at A with zero period 0 consumption. The straight line RS has the slope $-(1 + i)$ and it gives the terms on which consumption can be shifted between periods via bond market transactions. The optimum level of investment in the firm in period 0 is shown as $C_0^*\bar{C}_0$, such that RS is tangential to AB. The line AB has the slope $-(1 + \delta)$, where δ is the rate of return on investment for this firm. So, RS tangential to AB means that i is equal to δ. The firm invests up to the level where the rate of return is equal to the rate of interest.

Why is this the optimum? First note that if the owners invest so as to get to a, they can then borrow/lend via the bond market so as to end up with the consumption levels given by point b where RS is tangential to the consumption indifference curve *UU*. Now consider an investment decision that leads to a point to the right or the left of a along AB. Such a point will lie on a line parallel to but inside, beneath, RS. Moving along such a line so as to maximise utility, it will not be possible to get to as high a level of utility as that corresponding to *UU*.

The point here is that given the existence of the bond market, utility maximisation for the owners of firms involves two distinct steps. First, choose the level of investment in the firm so as to maximise its present value. Second, then use the bond market to borrow and lend so as to maximise utility. The present value of the firm is the maximum that its owners could borrow now and repay, with interest, from future receipts. In this two-period case, the firm's present value is $M_0 + [1/(1 + i)]M_1$, where M_0 and M_1 are receipts in periods 0 and 1, and $[1/(1 + i)]M_1$ is the 'discounted value' of M_1. Discounted values in a multiperiod context will be discussed below.

Now, this two-stage maximisation process applies to the owners of all firms. In each firm investment is undertaken up to the level where the rate of return is equal to the rate of interest. It follows that rates of return are equalised across firms, as is required by the second condition for efficiency in intertemporal allocation, equation 11.3.

We have seen that with a market for loanable funds, all consumption discount rates will be equal to the market rate of interest, and that all rates of return will equal the market rate of interest. It follows that the common consumption discount rate is equal to the common rate of return, as is required by the third condition, equation 11.4, for an intertemporally efficient allocation.

So, the conditions for intertemporal efficiency would be satisfied by an ideal system of markets that includes a market for loanable funds. In order for the conditions to be satisfied, that market – the bond market as the story was told here – is itself required to satisfy certain conditions. It must, for example, be a competitive market in the sense that all participants act as price-takers. As we emphasised in Chapter 4, the purpose of this kind of analysis is not to propagate the idea that actual market systems do bring about efficient outcomes. It is to define the conditions under which market systems would do that, and hence to support policy analysis. In fact, there are many markets for different classes of loanable funds, which satisfy the ideal conditions to varying degrees, and none do so fully.

11.1.4 Intertemporal modelling

The main purpose of this section is to relate the foregoing analysis to some standard models and issues. In this section we will be revisiting some of the topics and models, and using some of the notation, introduced in Section 3.5 – you may find it helpful to read that material again before proceeding here.

11.1.4.1 Optimal growth models

In much of the literature, including this book, the model used for looking at intertemporal allocation problems frequently involves just one individual at each point in time. For the two-period case, instead of the two utility functions $U^A = U^A(C_0^A, C_1^A)$ and $U^B = U^B(C_0^B, C_1^B)$, such models have the single function $W = W\{U(C_0), U(C_1)\}$. In such models, as well as aggregating over commodities and looking just at 'consumption', we are also aggregating over individuals and looking at a single 'representative' individual. The preference system represented by $W\{.\}$ has two components. $U(C_0)$ and $U(C_1)$ are contemporaneous utility functions which map consumption at a point in time into utility at a point in time.[1] It is assumed that $U(.)$ is invariant over time, and that it exhibits decreasing marginal utility. The function $W\{.\}$ maps a sequence of contemporaneous utility levels into a single measure for the whole sequence. In the literature, $W\{.\}$ is frequently given the particular form

$$W = U(C_0) + \left(\frac{1}{1+\rho}\right)U(C_1)$$

where ρ is the utility discount rate, a parameter, introduced in Chapter 3.

The function $W\{.\}$ can be, and is in the literature, interpreted in two ways. It can be treated as a particular functional form for the intertemporal utility function of a representative individual alive in both periods, one which is additively separable in discounted contemporaneous utilities. Alternatively, it can be treated as an intertemporal social welfare function where there are distinct, non-overlapping, generations alive in each period, each generation being represented by a single individual. We will, as we did in Chapter 3, use the first interpretation.

The way in which the function $W\{.\}$ is widely used in the literature is in 'optimal growth' models. In such models it is assumed that the conditions for efficiency in allocation are satisfied. Clearly, with just one commodity and one individual explicitly modelled there is little to be said about either intra-temporal or intertemporal efficiency. Note that where there are many individuals and commodities, efficiency requires equality across individuals' commodity consumption discount rates and across investment rates of return in the production of commodities. If it is assumed that these conditions are satisfied, working with a single commodity and

[1] As noted below, in most of the literature these sorts of problems are considered in continuous time, and in that setting the arguments of $W\{.\}$ are known as 'instantaneous' utilities. We switch to this terminology below when we move to continuous time.

a representative individual follows naturally. It is, then, the matter of the intertemporal distribution of utility, via saving and investment, that is investigated in such models. For our two-period case this investigation uses the problem of maximising

$$W = U(C_0) + \left(\frac{1}{1+\rho}\right)U(C_1) \tag{11.5a}$$

subject to the constraints

$$Q_0(K_0) - (K_1 - K_0) = C_0 \tag{11.5b}$$

$$Q_1(K_1) - (K_2 - K_1) = C_1 \tag{11.5c}$$

where Q_t is the output of the commodity during period t, and K_t is the capital stock at the beginning of period t. Assuming that capital is the only input to production further serves to simplify and sharpen the focus on the central issue, without the loss of anything essential. Note that the efficiency problem here is trivial. From 11.5b and 11.5c it is clear that no further conditions are required to ensure that consumption, and hence utility, in one period can only be increased at the cost of a reduction in consumption, and hence utility, in the other period.

Appendix 11.1 works through this intertemporal optimisation exercise and shows that

$$\frac{U_{C1}}{U_{C0}} = \frac{1+\rho}{1+\delta} \tag{11.6}$$

is a necessary condition for intertemporal welfare maximisation. For ρ less than δ, U_{C1} is less than U_{C0}, which for decreasing marginal utility means that C_1 is larger than C_0 – consumption is increasing over time. This makes sense, given that ρ measures the rate at which future utility is discounted, while δ measures the pay-off to deferring consumption and utility by investing. For ρ equal to δ equation 11.6 says that consumption would be the same in both periods.

Without the restriction to two periods, this kind of intertemporal welfare function becomes:

$$W = U(C_0) + \left(\frac{1}{1+\rho}\right)U(C_1) + \left(\frac{1}{1+\rho}\right)^2 U(C_2) + \ldots + \left(\frac{1}{1+\rho}\right)^T U(C_T) = \sum_{t=0}^{T}\left(\frac{1}{1+\rho}\right)^t U(C_t)$$

Most analysis of intertemporal distribution issues uses continuous time,

$$W = \int_{t=0}^{t=T} U(C_t)e^{-\rho t}dt$$

and frequently the time horizon is indefinitely far into the future so that

$$W = \int_{t=0}^{t=\infty} U(C_t)e^{-\rho t}dt \tag{11.7a}$$

The corresponding formulation of the constraint reflecting the possibilities for shifting consumption over time, equations 11.5b and 11.5c for the two-period model above, is

$$\dot{K} = Q(K_t) - C_t \tag{11.7b}$$

where $\dot{K}$ is the time derivative of K, i.e. the rate of investment. The maximisation of equation 11.7a subject to equation 11.7b is the basic standard optimal growth model. The mathematics of the solution to this maximisation problem are set out in Appendix 14.1. Corresponding to equation 11.6 above for the two-period case, for this continuous-time infinite-horizon version of the problem a necessary condition is:

$$\frac{\dot{U}_C}{U_C} = \rho - \delta \tag{11.8}$$

The left-hand side here is the proportional rate of change of marginal (instantaneous) utility, and along the optimal consumption path this is equal to the difference between the utility discount rate and the rate of return to investment. The former is a parameter, while the rate of return varies and is generally assumed to fall as the size of the capital stock increases. Given the assumption of diminishing marginal utility, $\delta < \rho$ implies that the left-hand side of equation 11.8 is negative which implies that consumption is growing along the optimal path. For $\delta = \rho$, growth is zero. Given the standard assumptions about the instantaneous utility and production functions, optimal growth for an intertemporal welfare function which adds discounted utilities takes the general form shown in Figure 11.7, which was previously seen as panel a of Figure 3.9 in Chapter 3.

In Part IV of this book we focus on models where natural resources are used, with capital and labour, in production. In the terminology introduced in Chapter 2, the natural resources that we shall be concerned with there are 'stock' resources, and are

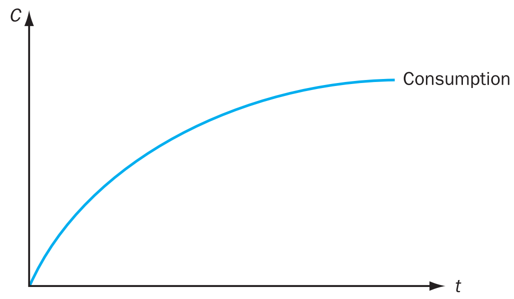


Figure 11.7 Optimal growth

potentially exhaustible. In analysing the use of such resources, attention focuses mainly on the patterns of use over time. An example of the sort of problem that you will be looking at in Part IV, and have already looked briefly at in Chapter 3, is the maximisation of

$$W = \int_{t=0}^{t=\infty} U(C_t)e^{-\rho t}dt \tag{11.9a}$$

subject to the constraints

$$\dot{K} = Q(K_t, R_t) - C_t \tag{11.9b}$$

$$\dot{S} = -R_t \tag{11.9c}$$

The first of the constraints says, as before, that output, Q, can either be used for consumption, C, or investment, $\dot{K}$. It differs from equation 11.7b in that the production of output now involves two inputs, capital, K, and some natural resource, R. In equation 11.9c, S stands for stock, and this constraint says that the natural resource being used is non-renewable.

This problem will be considered in some detail in Chapter 14, and variants of it – such as for the case of a renewable resource – will take up much of Part IV. The point that we want to make here is that whereas in the maximisation problem defined by equations 11.7 intertemporal efficiency is trivially guaranteed, in the case of equations 11.9 it is an essentially important feature of the problem. Notwithstanding that just one commodity is produced in the model of equations 11.9, there are two forms that investment can take. As before, current consumption can be forgone and output instead added to the capital stock. Additionally, there is now the possibility of reducing the current rate of use of the resource so as to leave more of it for future use. In terms of the analysis of the preceding sections there is now a role for an intertemporal efficiency condition. Inter-temporal efficiency requires the equalisation of the rates of return to capital accumulation and resource conservation.

11.1.4.2 Utility and consumption discount rates

Panel a of Figure 11.8 shows, as $W_U W_U$, a welfare indifference curve drawn in utility space for the intertemporal welfare function:

$$W = U(C_0) + \left(\frac{1}{1+\rho}\right)U(C_1)$$

Points along $W_U W_U$ are combinations of U in period 0 and U in period 1 that yield equal levels of W. $W_U W_U$ is a straight line with slope – (1 + ρ). Given that in each period U depends solely on that period's consumption, we can map $W_U W_U$ into $W_C W_C$, shown in panel b of Figure 11.8, the corresponding welfare indifference curve in consumption space.

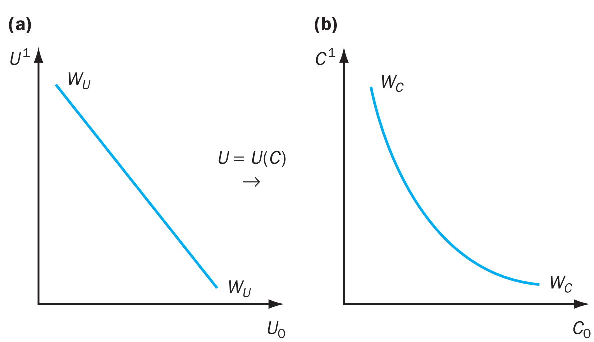


Figure 11.8 Indifference curves in utility and consumption space

In regard to Figure 11.1, we defined, for each individual, the consumption discount rate as the slope of the indifference curve in consumption space, multiplied by −1, minus 1. It is shown in Appendix 11.1 that the slope of $W_C W_C$ in panel b of Figure 11.8 is

$$\frac{U_{C0}}{[1/(1+\rho)]U_{C1}}$$

so that

$$r = \frac{U_{C0}}{[1/(1+\rho)]U_{C1}} - 1 = \frac{(1+\rho)U_{C0}}{U_{C1}} - 1 \qquad (11.10)$$

gives the relationship between the consumption rate of interest and utility discount rate for an intertemporal welfare function which is the sum of discounted contemporaneous utilities. Although derived here for the two-period case, this result holds generally. Also as shown in Appendix 11.1, working in continuous time, it can be established that

$$r = \rho + \eta g \qquad (11.11)$$

where η is the elasticity of marginal utility for the instantaneous utility function, and g is the growth rate for consumption.

From either 11.10 or 11.11, it can be seen that constant consumption implies that the consumption and utility discount rates would be equal. For consumption growing, the consumption discount rate is greater than the utility discount rate. In fact, unless η is zero, growing consumption would imply a positive consumption discount rate if the utility discount rate were zero. Discounting future consumption does not, that is, necessarily entail discounting future utility.

In the economics literature generally, and in the cost–benefit analysis literature particularly, when the terms ‘discounting’ or ‘discount rate’ are used without qualification, it is usually consumption discounting that is being referred to.

11.2 Project appraisal

In the preceding section of the chapter we set out some of the basic ideas of intertemporal welfare economics. In this section we are going to consider how those ideas are applied using CBA. While it is in the next section of the chapter that we will look specifically at CBA and the environment, Box 11.1 appears here so as to ‘set the scene’ for the material to be covered in this section – the analysis that it reports distinguishes between the commercial and the social appraisal of afforestation projects.

Box 11.1 A CBA of temperate-zone forestry

Is there an economic case for government support for afforestation programmes in temperate zones such as the UK? What are the benefits and costs of such programmes? David Pearce argues that afforestation programmes are multiple-output activities. The outputs he identifies are listed below.

- *T* Timber values
- *R* Recreational amenities
- *D* Biological diversity
- *L* Landscape values
- *W* Water-related effects: watershed protection, affecting soil erosion and water run-off, fixation of airborne pollutants, typically increasing pollutant concentrations locally but reducing them elsewhere
- *M* Microclimate effects
- *G* Carbon stores
- *S* Economic security
- *I* Community integration

Each of these outputs can be beneficial, relative to alternative uses of the land. However, in some cases the benefits may be negative. For example, if single-species spruce afforestation displaces the provision of wilderness areas, biological diversity is likely to diminish. On the other hand, the creation of urban forests in areas of industrial dereliction would, in most cases, increase diversity.

What are the costs of afforestation? These costs comprise land acquisition, planting, maintenance, thinning and felling. Denoting the present values for total benefits by B and total costs by C afforestation is economically justified if

$$B - C > 0$$

Pearce notes that only one of the joint products – the produced timber – is actually traded through market exchanges. All other products are beneficial (or sometimes adverse) external effects, not captured in market valuations.

Box 11.1 *continued*

On the other hand, the costs of afforestation are internalised in market transactions. The consequence of this is that afforestation programmes in temperate regions such as the UK are rarely commercially profitable. By way of example, Pearce quotes results from an earlier study. He introduces time into his analysis, discounts consumption-equivalent benefits and costs at a discount rate of 6%, and then estimates the net present value of various types of forestry plantations (on various types of land) under a variety of assumptions about the costs of land.

Pearce investigates eight types of forestry scheme. For each scheme, the commercial NPV is calculated under high and low (and sometimes zero) assumed costs of land. Of the 17 cases this generates, all but one result in negative NPVs. The sole exception is mixed fir/spruce and broadleaf plantations in lowlands, assuming the true value of land is zero (that is, the land has no alternative use).

Having evaluated the commercial returns to afforestation, Pearce then investigates each of the non-marketed benefits, so as to do an appraisal from a social viewpoint. He considers the benefits for each of the outputs *R*, *D*, *L*, *W*, *G*, *S* and *I*. For two of these (*R* and *G*) the benefits are quantified in money terms; for others (*D*, *W*, *S* and *I*) Pearce identifies and describes the benefits but does not attempt any monetary quantification. Unquantified benefits will have to be judgementally taken into account when project decisions are made.

Recreational benefits for various forms of afforestation are taken from Benson and Willis (1991). The gross values for recreational benefits in the UK range from £3 per hectare on low-amenity woodlands in the uplands to £424 per hectare on very high-amenity lowland woodlands (in 1989 prices). Pearce suggests these values are likely to grow in real terms by at least 1% per annum. Wildlife conservation and biodiversity benefits (*W*) and landscape amenity values (*L*) are two outputs that Pearce does not quantify and monetise. He argues that these benefits will vary widely depending upon woodland form and location, but that they are likely to be positive in the UK, where land for afforestation tends to be drawn from low-wildlife-value agricultural land. However, if afforestation takes the form of non-native conifer species, and is at the expense of previously semi-natural land use, these effects on both landscape amenities and biological diversity could be strongly negative. The picture is thus a very mixed one, with the magnitude (and direction) of the effects varying greatly from one case to another.

Table 11.1 Forestry classes that pass the CBA test

Forest type	Assumptions giving positive NPV at $r = 6\%$
FT5 Community forests	Very high recreational values
FT4 Spruce in uplands	Moderate recreational values and land values at $0.5 \times$ market price
FT8 Fir, spruce, broadleaf trees in lowlands	High recreational values and land values at $0.8 \times$ market price
FT7 Pine in lowlands	Moderate recreational values and land values at $0.5 \times$ market price

Source: Adapted from Pearce (1994)

Water-related ecological outputs (*W*) discussed by Pearce include the effects of afforestation on water supply, water quality, the deposition of air pollution, soil erosion, and the impacts of fertiliser and pesticide use and harvesting practices. Qualitative estimates only are presented for these impacts.

Greenhouse-warming-related effects are quantified in monetary terms by Pearce. His estimates of the present value of benefits from carbon fixing, in pounds per hectare at a 6% discount rate, range from £142 on upland semi-natural pinelands to £254 on lowland mixed woodlands.

Pearce's conclusions

Adding to the commercial benefits net of commercial costs the estimates for the two social benefit categories that he was able to quantify (recreation and carbon-fixing), Pearce concludes that four of the eight general classes of woodlands he investigates have a clear social justification for increased afforestation at a discount rate of 6%. In the four categories considered in Table 11.1, that is, increased afforestation passes the CBA test, though it fails on a commercial test.

As explained above, these conclusions are drawn without looking at non-monetised benefits (or costs). In those cases where the (social) NPV of an afforestation project is negative, the decision maker may nonetheless regard the project as socially desirable if she forms a judgement that the non-monetised benefits are sufficiently large to offset the negative (monetised) NPV.

Source: Adapted from Pearce (1994)

We begin this section by looking at project appraisal as it would be conducted by a private-sector agent according to commercial criteria. This provides a useful way into CBA in terms of the principles and practice involved.

11.2.1 Private appraisal

The commercial viability of a project can be assessed in two equivalent ways – the net present value test and the internal rate of return test. Since the rationale is clearer in the former, we begin by looking at that.

11.2.1.1 The net present value test

At the interest rate i, £1 lent for one year grows to £$(1 + i)$. If at the end of the year, the principal and the interest earned are re-lent – left to accumulate in a savings account say – then after 2 years the amount due will be £$\{(1 + i)(1 + i)\}$ = £$(1 + i)^2$. After being lent for 3 years, the amount due will be £$\{(1 + i)(1 + i)(1 + i)\}$ = £$(1 + i)^3$. And so on and so on. This is the process of compounding.[2] Generally, a principal lent at the rate i, with annual compounding, will be worth V_t after t years where

$$V_t = \mathrm{PV}(1 + i)^t \tag{11.12}$$

where PV stands for the principal, the sum initially lent, or 'invested'.

What would a completely reliable promise to pay £$(1 + i)$ a year hence be worth now? In the previous section we called such a promise a 'bond'. Then, the question is: what is the value today of a bond with value £$(1 + i)$ a year from now? Given that £1 invested today at i will be worth £$(1 + i)$ a year from now, the answer to this question is clearly £1. This process of converting future amounts to current equivalents is discounting, which is compounding in reverse. Just as compounding can be extended over many years, so can discounting. What would be the value of a bond that promised to pay £V_t years from now? The answer is the amount of money that would have to be invested now at the ruling interest rate to realise £V_t years from now. By equation 11.12 that is

$$\mathrm{PV} = V_t/(1 + i)^t \tag{11.13}$$

where PV stands for 'present value' in the terminology used when looking at things this way round, and $1/(1 + i)^t$ is the discount factor for t years at interest rate i.

The present value of a sum of money in the future is its current equivalent, where equivalence is in the sense that, given the existence of facilities for lending and borrowing, an individual or firm would currently be indifferent between the certain promise of the future sum and the offer of the present value now.

Project appraisal is the consideration of whether it makes sense to make some expenditure commitment now given the expectation of future receipts as a result. Consider a simple example. Suppose that a firm can buy a machine for £100 now. If it does this, it can use the machine for some time, and its use will give rise to additional receipts of £50 for each of the two following years, and then of £45.005 the year after that. Then, the machine will be useless, and its scrap value 0. Using the machine will add £10 each year to costs. The impact of acquiring the machine on the firm over time is given by Table 11.2.

Table 11.2 Example net cash flow 1

Year	Expenditure	Receipts	Net cash flow
0	100	0	−100
1	10	50	40
2	10	50	40
3	10	45.005	35.005
4	0	0	0

Should the firm buy the machine? Summing the net cash flow over time gives a positive number £15.005. However, as is typical with investment projects, there is a negative cash flow now and the

[2] Compounding proceeds according to exponential growth. An interesting question is how long it takes for something growing exponentially, like an untouched savings account, to double in size. From equation 11.12, $V_t/\mathrm{PV} = (1 + i)^t$ so to find the doubling time solve $2 = (1 + i)^t$ or $\ln 2 = t \cdot \ln(1 + i)$ for t. For $i = 0.015$, for example, the doubling time is 47 years, and for $i = 0.03$ it is just 24 years. For $0 < i \le 0.075$ $\ln(1 + i)$ is approximately equal to i, and $\ln 2$ is close to 0.7, so an approximation to the doubling time is given by 0.7 divided by i.

Table 11.3 Example net cash flow 2

Year	Expenditure	Receipts	Net cash flow
0	100	0	−100
1	0	0	0
2	0	0	0
......	......	......	......
......	......	......	......
49	0	0	0
50	0	115.005	115.005
51	0	0	0

positive cash flow is in the future. Just looking at the total over the life of the project ignores this time profile. The net present value approach to project appraisal is a technique for assessing projects which takes account of the futurity of the positive elements in the net cash flow. It can be thought of as a way of normalising the cash flows associated with projects so that alternatives can be properly compared. To see the need for such normalisation, suppose that the firm considering the project described above could alternatively now invest £100 in a project which would give rise to the net cash flow shown in Table 11.3.

For both of these projects, the total net cash flow over their lifetimes is £15.005. On this basis the firm would be indifferent between the two projects, which clearly does not make sense.

The net present value, NPV, of a project is the present value of the net cash flow associated with it. If an investment has a non-negative NPV, then it should be undertaken, otherwise not. The decision rule is, that is, go ahead with the project only if NPV ≥ 0. The rationale for this rule is that following it will lead to going ahead only with projects that leave unchanged or increase net worth. A firm wishing to maximise its net worth should rank available projects by NPV, and undertake those for which NPV ≥ 0.

Denote expenditure in year t as E_t, and receipts as R_t so that $N_t = R_t - E_t$ is the net cash flow in year t, and denote the project lifetime by T. Then the present value of expenditures is

$$\mathrm{PV}_E = E_0 + \frac{E_1}{1+i} + \frac{E_2}{(1+i)^2} + \ldots + \frac{E_T}{(1+i)^T} = \sum_0^T \frac{E_t}{(1+i)^t} \tag{11.14}$$

the present value of receipts is

$$\mathrm{PV}_R = R_0 + \frac{R_1}{1+i} + \frac{R_2}{(1+i)^2} + \ldots + \frac{R_T}{(1+i)^T} = \sum_0^T \frac{R_t}{(1+i)^t} \tag{11.15}$$

and the net present value of the project is

$$\mathrm{NPV} = \mathrm{PV}_R - \mathrm{PV}_E = \sum_0^T \frac{R_t}{(1+i)^t} - \sum_0^T \frac{E_t}{(1+i)^t} \tag{11.16}$$

which can also be written

$$\mathrm{NPV} = N_0 + \frac{N_1}{1+i} + \frac{N_2}{(1+i)^2} + \ldots + \frac{N_T}{(1+i)^T} = \sum_0^T \frac{N_t}{(1+i)^t} \tag{11.17}$$

Applying 11.16 or 11.17 to the data of Table 11.2 gives

(i) for $i = 0.05$, NPV = £4.6151
(ii) for $i = 0.075$, NPV = £0
(iii) for $i = 0.10$, NPV = −£4.27874

so that while the project fails the NPV test at 10%, it passes at 7.5% and 5%, and has a higher NPV for an interest rate of 5% than it would for an interest rate of 7.5%. A close examination of how these results arise demonstrates the logic and meaning of the NPV test. This is made clearer if it is assumed that the firm finances the project by issuing one-year bonds.

Take the 5% case first. In order to acquire the machine, the firm must on day one of year 0 sell its bonds to the value of £100. Given $i = 0.05$, it thus incurs the liability to redeem the bonds for £105 on day one of year 1. At that time, it will have net receipts from using the machine of £40, a shortfall of £65. It covers this shortfall by issuing new bonds in amount £65, which generates a liability of £68.25 (65 × 1.05) for day one of year 2. At that time its receipts in respect of using the machine are £40, so there is a shortfall of £28.25 as between net receipts and expenditure on bond redemption. This can be covered by issuing further one-year bonds to the value of £28.25, incurring a liability of £29.6625 (28.25 × 1.05) for day one of year 3. On that day, net receipts will be 35.005, so that there will be a current surplus of 35.005 – 29.6625 = £5.3425 at the end of the project lifetime. What is the present value of this surplus when considered at the time, day one of year 0, that a decision has to be made on the project? It is $5.3425 \times 1/(1 + i)^3 = 5.3425/1.1576 = £4.6151$, which is the answer given by the NPV formula for this project with an interest rate of 5%, see (i) above. The NPV of a project is the amount by which it increases net worth in present value terms.

Working through the 7.5% case in the same way –

t
0 sell £100 of bonds
1 redeem bonds for £107.5, sell £67.5 of new bonds (107.5 – 40)
2 redeem bonds for £72.5625, sell £32.5625 of new bonds (72.5625 – 40)
3 redeem bonds for £35.005, surplus of £0

and 11.16 or 11.17 produces the answer NPV = 0.

For the 10% case

t
0 sell £100 of bonds
1 redeem bonds for £110, sell £70 of new bonds (110 – 40)
2 redeem bonds for £77, sell £37 of new bonds (77 – 40)
3 redeem bonds for £40.7, surplus of –£5.695 (35.005 – 40.7)

and the formula produces the answer NPV = –£4.27874. This is the present value at 10% of –£5.695 three years hence. £4.27874 is what would have to be initially invested at 10% to yield enough to meet the £5.695 liability that would arise after three years if the firm went ahead with this project.

Projects with positive NPV increase net worth, while those with negative NPV reduce it. If the NPV is 0, the project would leave net worth unchanged.

In this example, for a given project net cash flow, a higher interest rate means a lower NPV. It is sometimes assumed that this is always true. It is not. It is true where the time profile is one with negative net receipts early followed by positive net receipts. But, and this can be important where projects have long-term environmental impacts, when proper account is taken of all costs and benefits the time profile may not be like this.

Table 11.3 provided the data for an alternative project, which involved £100 expenditure now for a one-off net receipt of £115.005 in year 50. For this project with 5% interest rate

$$\begin{aligned} NPV &= \{115.005/1.05^{50}\} - 100 \\ &= \{115.005/11.4\} - 100 \\ &= 10.0289 - 100 \\ &= -£89.9711 \end{aligned}$$

Both projects, Tables 11.2 and 11.3, have total lifetime net cash flows of £15.005. But, whereas, at 5%, the Table 11.2 project has a positive NPV, the Table 11.3 project has a large negative NPV, reflecting the 50-year wait for a positive cash flow.

The logic of the NPV test for project appraisal has been developed here for a situation where the firm is going to borrow the funds to finance the project, as this makes clearer what is going on. However, the test is equally appropriate where the firm can fund the project from its own cash reserves. This is because the firm could, instead of using its own cash to finance the project, lend the money at the market rate of interest. If the NPV for the project is negative, the firm would do better for the present value of its net worth by lending the money rather than committing to the project. If the NPV is 0, it is a matter of indifference. If the project has a positive NPV, then the money would do more for the present value of net worth by being put into the project than being lent at interest.

Where the project lifetime is more than a few years, finding the NPV from data on the projected net cash flow is straightforward but tedious. Most spreadsheet software includes a routine that calculates NPV, and the internal rate of return which we now discuss.

11.2.1.2 The internal rate of return test

An alternative test for project appraisal is the internal rate of return, IRR, test, according to which a project should be undertaken if its internal rate of return is greater than the rate of interest. The internal rate of return for a project is the rate at which its net cash flow must be discounted to produce an NPV equal to 0.

Recall that NPV is given by:

$$\text{NPV} = N_0 + \frac{N_1}{1+i} + \frac{N_2}{(1+i)^2} + \ldots + \frac{N_T}{(1+i)^T}$$
$$= \sum_0^T \frac{N_t}{(1+i)^t}$$

A project's IRR is found by setting the left-hand side here equal to zero, and then solving the equation for the interest rate, which solution is the IRR. The IRR is, that is, the solution for x in

$$0 = N_0 + \frac{N_1}{1+x} + \frac{N_2}{(1+x)^2} + \ldots + \frac{N_T}{(1+x)^T}$$
$$= \sum_0^T \frac{N_t}{(1+x)^t} \qquad (11.18)$$

The IRR test will, for the same input data, give the same result as the NPV test. The reason for this, and the underlying logic of the IRR test, is apparent from the discussion of the NPV test. In some cases, because of the time profile of the net cash flow, the solution to 11.18 involves multiple solutions for x. This problem does not arise with the NPV test, and it is the recommended test.

11.2.1.3 Dealing with risk

Thus far it has been assumed that at the time of appraising a project, the firm knows what the cash flows that it would give rise to are. This, of course, is generally not the case. The net cash flow figures that are input to NPV or IRR calculations are derived from projections, or estimates, of future receipts and expenditures, and an important question is: how do we incorporate into project appraisal the fact that it is dealing with imperfect knowledge of the future?

If the firm is prepared to assign probabilities to possible alternatives regarding the determination of the net cash flow, a simple modification of the NPV criterion can be used. Instead of requiring that the NPV be positive, it is required that the expected NPV be positive. The expected value, or expectation, of a decision is the probability weighted sum of the values of the mutually exclusive outcomes. Suppose that for the project considered above, instead of the single known net cash flow considered thus far, the firm considers that there are two possible outcomes with the probabilities shown in Table 11.4.

Table 11.4 One project, two possible cash flows

Year	Net cash flow 1 Probability 0.6	Net cash flow 2 Probability 0.4
0	−100	−100
1	40	35
2	40	35
3	35.005	25
4	0	0

Table 11.5 shows the calculations to calculate the expected NPV. In this case it is negative and the project should not be undertaken.

Basing the decision rule on the expected NPV assumes that the decision maker is 'risk-neutral', which means that she regards an expected value of £x as the same as the certainty of £x. Thus, a risk-neutral decision maker would regard the offer of £4 if a tossed coin comes up heads, where the expected value of the offer is $(0.5 \times 4) + (0.5 \times 0) =$ £2, as equivalent to the offer of £2 cash in hand. Decision makers are in fact frequently observed to be 'risk-averse' rather than risk-neutral, e.g. would prefer £2 cash in hand to £4 if heads comes up. There are a variety of ways to modify the basic NPV decision rule to deal with decision makers who are risk-averse. References to the literature are provided in the Further Reading section at the end of the chapter. We will revisit the question of imperfect knowledge of project consequences in Chapter 13. Our discussion of CBA in this chapter will, in the main, assume that project consequences are known.

Table 11.5 Calculation of expected NPV

Year	Expected net cash flow	Present value of expected cash flow
0	$-(0.6 \times 100) + \{-(0.4 \times 100)\} = -100$	−100
1	$(0.6 \times 40) + (0.4 \times 35) = 38$	$38/1.075 = 35.35$
2	$(0.6 \times 40) + (0.4 \times 35) = 38$	$38/1.075^2 = 32.88$
3	$(0.6 \times 35.005) + (0.4 \times 25) = 31.003$	$31.003/1.075^3 = 24.96$
4	$(0.6 \times 0) + (0.4 \times 0) = 0$	
Expected NPV		−6.81

A flexible way of informally considering the impact of risk would be to compute the NPV for different assumptions about future expenditures and receipts, to examine the sensitivity of the decision to assumptions built into the net cash flow projections. This kind of sensitivity analysis does not produce a unique decision, but it can illuminate key areas of the underlying project analysis.

Table 11.6 Net benefit (NB) impacts consequent upon an illustrative project

	Time period				
Individual	0	1	2	3	Overall
A	$NB_{A,0}$	$NB_{A,1}$	$NB_{A,2}$	$NB_{A,3}$	NB_A
B	$NB_{B,0}$	$NB_{B,1}$	$NB_{B,2}$	$NB_{B,3}$	NB_B
C	$NB_{C,0}$	$NB_{C,1}$	$NB_{C,2}$	$NB_{C,3}$	NB_C
Society	NB_0	NB_1	NB_2	NB_3	

11.2.2 Social project appraisal

CBA is the social appraisal of projects. It is used for appraising public-sector projects, including policies, and private-sector projects where some of the consequences of going ahead with the project would not get market prices attached to them, i.e. would involve external effects in the terminology of Chapter 4. Where there are external effects, as is the case with (but not only with) many environmental impacts, project appraisal using market prices would mean completely ignoring those consequences. In such circumstances, a social appraisal procedure, CBA, is required to assess the project properly from a social, as opposed to private, commercial, perspective. The basic idea in CBA is to correct for market failure due to externalities in assessing a project's costs and benefits. Typically a CBA would be carried out by, or for, a public-sector agency. In this section we will, except where stated otherwise, assume for the purposes of discussion that we are looking at a public-sector investment project.

CBA uses the NPV test. If, after correcting for market failure by taking externalities into account, by attaching monetary valuations to them in ways to be discussed in Chapter 12, a project has a positive NPV, then it should go ahead. There are two ways of coming at this. We look first at an interpretation of the NPV test in CBA that is a fairly natural extension of the way it works for private sector project appraisal. Then we look at an interpretation in terms of social welfare enhancement. The latter is now the more prevalent approach to CBA.

Whichever interpretation is being followed, it is important to be clear that the first stages in any CBA are to properly assess the capital cost of the project, and to forecast all of the consequences of going ahead with it for each and every affected individual in each year of the project's lifetime. In what follows here we pay little attention to these matters, but clearly they are important. It is a basic assumption of CBA that all of the consequences for individuals can be expressed in terms of monetary gains and losses. To the extent that this cannot be done for a project, the CBA is incomplete, and would have to be treated as indicative rather than definitive, and clearly reported as such. The case reported in Box 11.1 exemplifies this. In what follows here, we shall assume that complete monetarisation is possible.

Table 11.6 shows the results of this first stage for an illustrative project undertaken in period 0, when the capital cost is incurred, and affecting three individuals over three subsequent periods. NB stands for Net Benefit, the difference between gains and losses after correcting for market failure, measured in monetary units (£s, $s or whatever).

Our discussion of CBA is concerned with how to use the information in Table 11.6 to decide whether, according to welfare economics criteria, the project is socially desirable, and should go ahead. Appraisal involves, first, adding net benefits across individuals at a point in time to get contemporaneous net benefits $NB_0, \ldots, NB_3$, where $NB_t = NB_{A,t} + NB_{B,t} + NB_{C,t}$. The NPV of this project is then the discounted sum of net benefits:

$$NPV = NB_0 + \frac{NB_1}{1+r} + \frac{NB_2}{(1+r)^2} + \frac{NB_3}{(1+r)^3}$$

The decision rule is to go ahead with the project if its NPV is positive. Generally, for T periods, the project should go ahead if:

$$NPV = \sum_{t=0}^{t=T} \frac{NB_t}{(1+r)^t} > 0 \tag{11.19}$$

As noted, there are in the literature two rationales for this decision rule, and we now consider these.

11.2.2.1 CBA as a potential Pareto improvement test

The first follows immediately from the discussion above of private-sector appraisal, where the point is that a positive NPV indicates that, with due allowance for the dating of costs and benefits, the project delivers a surplus of benefit over cost. The consumption gains involved are, that is, greater than the consumption losses, taking account of the timing of gains and losses. The existence of a surplus means that those who gain from the project could compensate those who lose and still be better off. On this view, the NPV test in CBA is an intertemporal variant of the potential compensation, or potential Pareto improvement, test, which was discussed for the static setting in Chapter 4. It does not require that compensation is actually paid.

We can see what is involved using the two-period framework and notation from the first part of this chapter – the initial investment is ΔI_0, equal to $-\Delta C_0$, and the consumption increment on account of going ahead with the project is ΔC_1. The government can fund the project either by taxation or borrowing. In the former case first period consumers lose an amount ΔC_0, equal to ΔI_0, and second period consumers gain ΔC_1, and the question is whether the gain exceeds the loss. From the viewpoint of the first period, the second period gain is worth $\Delta C_1/(1 + r)$, so the question is whether

$$\Delta C_1/(1 + r) > \Delta I_0 \tag{11.20}$$

is true, which is the NPV test discounting at r.

If the government funds the project by borrowing and the public-sector project displaces, or crowds out, the marginal private sector project with rate of return δ, things are different. In this case the cost of the public sector project is ΔI_0 in the first period plus $\delta\Delta I_0$ in the second, this being the extra consumption that the private-sector project would have generated in the second period. In this case, from the viewpoint of the first period, the gain exceeds the loss if:

$$\frac{\Delta C_1}{1 + r} > \Delta I_0 + \frac{\delta\Delta I_0}{1 + r} \tag{11.21}$$

This is the NPV test with the consumption gain discounted at the consumption rate of interest, and compared with the cost of the project scaled up to take account of the consumption that is lost on account of the displaced private-sector project.

11.2.2.2 CBA as a welfare increase test

Recall from Chapter 4 that interest in compensation tests in welfare economics derives largely from the rejection of the idea that utility can be measured cardinally. If only ordinal measurement is possible, then we cannot meaningfully aggregate individual utilities, and cannot properly construct social welfare functions, though we can consider questions about efficiency. Interpreting CBA as a potential Pareto improvement test means it is consistent with assuming only ordinal utility measurement. While some economists are content to treat CBA as a means for pursuing efficiency objectives, others are willing to assume that utilities can be aggregated via social welfare functions, and approach CBA on that basis.

On this approach, rather than start with the illustrative project data as laid out in Table 11.6, we start with Table 11.7, where, for example, $\Delta U_{B,2}$ denotes the change in utility during time period 2 that would be experienced by individual B on account of the project if it went ahead. If there existed a generally agreed social welfare function with dated individual (cardinal) utilities as arguments, the analyst could compute

$$\Delta W = W(\Delta U_{A,0}, \ldots, \Delta U_{C,3})$$

and consider its sign. If positive the project should go ahead. Alternatively, we could imagine that there existed an intratemporal social welfare function which mapped individual utilities into a social aggregate, ΔU_t, in each period, and an intertemporal social welfare function for aggregating over time. The analyst would then compute

$$\Delta W = W(\Delta U_0, \Delta U_1, \Delta U_2, \Delta U_3)$$

Table 11.7 Changes in utility (ΔU) consequent on an illustrative project

	Time period				
Individual	0	1	2	3	Overall
A	$\Delta U_{A,0}$	$\Delta U_{A,1}$	$\Delta U_{A,2}$	$\Delta U_{A,3}$	ΔU_A
B	$\Delta U_{B,0}$	$\Delta U_{B,1}$	$\Delta U_{B,2}$	$\Delta U_{B,3}$	ΔU_B
C	$\Delta U_{C,0}$	$\Delta U_{C,1}$	$\Delta U_{C,2}$	$\Delta U_{C,3}$	ΔU_C
Society	ΔU_0	ΔU_1	ΔU_2	ΔU_3	

and the decision would be based on the sign here. A widely entertained particular form for the intertemporal social welfare function, considered in the previous section, is

$$\Delta W = \Delta U_0 + \frac{\Delta U_1}{1+\rho} + \frac{\Delta U_2}{(1+\rho)^2} + \frac{\Delta U_3}{(1+\rho)^3}$$

where aggregation over time involves exponential utility discounting. If ΔW here is positive, the project should go ahead.

The problem with all of this is that the individual utility variations consequent upon going ahead with a project are not generally regarded as something that could be estimated *ex ante*, or observed *ex post*. The way forward is to take an individual's utility to be a function of her total consumption, and to equate individual net benefit to the change in an individual's total consumption. If a project causes an individual to suffer a reduction in utility, that loss is expressed in monetary terms, by the methods to be considered in the next chapter, and treated as a consumption loss for the individual. Similarly for gains. Adding across losses and gains for the individual gives her total consumption change, or net benefit, due to the project. This means that it is possible to use the project data of Table 11.6.

Given this step, the welfare enhancement appraisal can be conducted using the consumption change, or net benefit, data of Table 11.6. There are two dimensions involved here, the intra- and the inter-temporal. First, it has been suggested that, in terms of Table 11.6, contemporaneous total net benefit should be defined as the weighted sum of individual net benefits, with marginal utilities of consumption as weights, rather than as the simple sum. That is, using

$$\mathrm{NB}_t = U_C^{\mathrm{A}}\mathrm{NB}_{\mathrm{A},t} + U_C^{\mathrm{B}}\mathrm{NB}_{\mathrm{B},t} + U_C^{\mathrm{C}}\mathrm{NB}_{\mathrm{C},t}$$

where U_C^i is the ith individual's marginal utility of consumption, instead of

$$\mathrm{NB}_t = \mathrm{NB}_{\mathrm{A},t} + \mathrm{NB}_{\mathrm{B},t} + \mathrm{NB}_{\mathrm{C},t}$$

This suggestion is rarely followed in practice. It would require identifying the individuals, or groups of individuals, affected by the project, and then ascertaining the marginal utilities for those individuals or groups.

Second, there is the matter of aggregation over time. A social welfare function defined over utilities which are functions of contemporaneous consumption levels implies a social welfare function defined over those consumption levels. As discussed in section 11.1.4.2 above, and see also Appendix 11.1,

$$W = \sum_{t=0}^{T}\left(\frac{1}{1+\rho}\right)^t U(C_t)$$

implies

$$W = \sum_{t=0}^{T}\left(\frac{1}{1+r}\right)^t C_t \tag{11.22}$$

where

$$r = \frac{(1+\rho)U_{Ct-1}}{U_{Ct}} - 1$$

which in continuous time can be written as

$$r = \rho + \eta g \tag{11.23}$$

previously seen as equation 11.11. According to equation 11.22, welfare is a weighted sum of consumption at different dates, and it follows that the change in welfare is the same weighted sum of changes in consumption, i.e. aggregate net benefits.

Given this, the NPV test is interpreted as a test that identifies projects that yield welfare improvements. Positive and negative consumption changes, net benefits that is, are added over time after discounting, so that

$$\Delta W = \sum_{t=0}^{T}\left(\frac{1}{1+r}\right)^t \Delta C_t \tag{11.24}$$

and for $\Delta W > 0$ the project is welfare enhancing and should go ahead.

Again, it is useful to look at the basic idea over two periods, and to consider project financing. If the government finances the project out of current tax receipts, then from equation 11.24 we have:

$$\Delta W = -\Delta I_0 + \left(\frac{1}{1+r}\right)\Delta C_1 \tag{11.25}$$

The right-hand side here is just the project's NPV, which is positive if

$$\Delta C_1/(1+r) > \Delta I_0$$

which is the condition 11.20 above.

If the project is financed by borrowing, and it crowds out the marginal private-sector project, then, again, allowance has to be made in assessing the cost of the project for the lost return on that private-sector project. In that case

$$\Delta W = -\Delta I_0 + \left(\frac{\Delta C_1}{1+r} - \frac{\delta \Delta I_0}{1+r}\right) \tag{11.26}$$

where the right-hand side is the NPV for the project when account is taken of the return on the displaced private-sector investment. Here ΔW will be positive if

$$\frac{\Delta C_1}{1+r} > \Delta I_0 + \frac{\delta \Delta I_0}{1+r}$$

which is the same as the condition 11.21.

This approach to the rationale for CBA, regarding it as a test for a welfare gain, is now the dominant one in the public sector and welfare economics literature. If the NPV is positive, and we assume cardinal utility and the social welfare functions implied, we can use 11.24 to say that the project would increase welfare. If we do not want to assume cardinal utility, we can use 11.20 to say that the project delivers a social surplus. In either case, we are assuming that the project consequences included in the appraisal can be properly expressed in monetary equivalent terms for the affected individuals.

11.2.3 Choice of discount rate

There are a number of technical aspects of the application of CBA that warrant extended discussion. One of these is the means by which the correct, i.e. market-failure-correcting, contemporary monetary valuations are assigned to project impacts. We devote the whole of the next chapter, and some of the one after that, to this topic, albeit exclusively in regard to environmental impacts. Space precludes dealing with the other issues properly: see the Further Reading suggestions at the end of the chapter. However, we will discuss here the question of the discount rate to be used in CBA, because it is important, and can be a source of confusion if care is not taken.

There has been, and is, disagreement among economists about the discount rate to be used, for a given economy, in CBA as illustrated in Box 11.2. This is important because the decision reached on a project using the NPV test can be very sensitive to the number used for the discount rate. This is particularly the case where, as with many projects involving environmental impacts, the time horizon for the NPV test is many years into the future. In this connection it is important to note that the proper time horizon for the appraisal of a project is the date at which its impacts cease, not the date at which it ceases to serve the purpose for which it was intended. Thus, for example, for a nuclear fission plant the time horizon is not the 40 years to the time when it ceases to generate electricity but the time over which it is necessary to devote resources to storing the plant's waste products – hundreds of years.

Table 11.8 Present values at various discount rates

	Time horizon			
	Years			
Discount rate %	25	50	100	200
0.5	88.28	77.93	60.73	36.88
2	60.95	37.15	13.80	1.91
3.5	42.32	17.91	3.21	1.03
7	18.43	3.40	0.12	0.0001

Table 11.8 gives the present value of £100 arising from 25 to 200 years ahead at discount rates from 0.5% to 7%. This range of discount rates is not arbitrary – it is that found in Box 11.2. Clearly, the choice of discount rate matters. Even for 25 years out, the present value at 2% is three times that at 7%.

We should note that there is in the literature complete agreement on one thing. Either the entire CBA should be done for a constant general price level together with a real discount rate, or it should be done for current prices together with the nominal discount rate. In practice, it is almost always the first of these that gets done. The figures cited in Box 11.2 are all for real rates. A nominal rate can be derived from a real rate using the inflation rate. For the levels usually experienced for real interest and inflation rates, to a close approximation the nominal rate is the real rate plus the rate of inflation.[3]

In the first section of this chapter we introduced and considered r the consumption rate of interest, δ

[3] With r for the real rate, n for the nominal rate and p for the rate of inflation, the exact relationship is $n = r + p - rp$. With, for example, a real rate of 0.03 and an inflation rate of 0.04, this gives 0.0688 as compared with the approximate result of 0.07.

the marginal rate of return on investment, and i the market rate of interest. We showed that in ideal circumstances, with no intertemporal market failure, we would have $r = i = \delta$. In that case there would be no choice to be made as between r, i and δ. In the real world, however, these three rates are not equal, and economists have argued over which rate should be used to do the NPV test in CBA. Should it be the market rate of interest i, or the consumption discount rate r, or the marginal rate of return δ. Further, if there were agreement about using one of these, how in practice should it be fixed at a particular numerical value?

The reader may have noticed that we have already answered the first of these questions implicitly. In our discussions of CBA in this section, only r has ever appeared as a discount rate in the various equations and expressions. It is now widely agreed that, because of the logic as we have set it out here, the proper rate of discount in CBA is the consumption discount rate. This is true whether we want to treat CBA as a potential Pareto improvement test, or as a test for a positive welfare change.

While it is true that only r appears as a discount rate, the reader will have noticed that δ appears in 11.21 and 11.26, which relate to the NPV test when the government finances the project by borrowing. What these expressions indicate is that we should discount at the consumption rate *and* adjust the initial cost of the project for the fact that it displaces a private-sector project with rate of return δ. Doing the latter is known in the literature as 'shadow pricing' the capital input. In this two-period illustration, each actual £'s worth of capital expenditure on the project goes into the CBA as £1 plus £(δ/1 + r), so that the capital costs are increased by an amount depending on the values for δ and r. Where the project lifetime is more than two periods, the expression for the shadow price for capital is more complicated, and depends on T (project lifetime) as well as δ and r. We will not go into this here, for reasons now to be explained (but the interested reader can follow capital shadow pricing up in references given in Further Reading at the end of the chapter).

Until the early 1990s the dominant view among economists was that the proper way to do the CBA of public-sector projects where borrowing was involved, as would usually be the case, was to shadow price the capital inputs to the project, and then to discount the net benefits using r, the consumption discount rate. This made life quite difficult because in practice working out the proper shadow price could be complicated. Basically, to do it properly would require working out the consequences of government borrowing for future consumption flows for each project. This would depend on such things as the private propensity to save and the rate of taxation on capital-based income. The question of the proper shadow pricing of capital received quite a lot of attention in the public finance literature.

The validity of the crowding-out assumption was called into question in the early 1990s. It was argued that, for advanced market economies anyway, given the international capital mobility that was by then the norm, it would be more appropriate to assume no crowding out than to assume 100% as above. It was argued, that is, that given international capital mobility, the supply of capital for private-sector projects should be treated as perfectly elastic. This view has become the predominant view, and, in market economies open to the international capital market, the shadow pricing of capital is not now seen as necessary in CBA. This is why we do not go into the complexities of capital shadow pricing here.

This means that the majority recommendation now is just to discount project net benefits at the consumption rate, r.

11.2.3.1 Where to get a number for r?

So, whether we want to look at the NPV test in the CBA of public-sector projects as a potential compensation test or a test for a positive welfare change, we end up at the same position for actually doing the test – work out the future flows of net benefits and discount those using the consumption rate of discount.

The fact that both rationales for the NPV test in CBA lead to the same actual test does not mean that making the distinction between them is entirely redundant. This is because which rationale is the starting point has some bearing on what is considered to be the appropriate way of getting a number for r to use in CBA. In Chapter 3 when discussing ethics and welfare economics we made a distinction between the prescriptive and descriptive approaches to the question of the appropriate value for the utility discount rate, ρ. A similar distinction is useful in understanding the differing arguments in the literature about how to arrive at a value for the consumption discount rate for use in CBA. To some extent, the distinction here aligns with that between the view that the NPV test is a potential compensation test and the view that it is to identify projects for which ΔW is positive.

Coming at things from the potential Pareto improvement perspective, it would be natural to adopt the descriptive position, and ask 'what *is* the consumption discount rate?'. On this basis, the CBA is concerned only with efficiency considerations, so there is no need to consider matters that relate to the distribution of gains and losses as between individuals and over time. It is generally considered that the actual consumption discount rate can be identified with the post-tax return on risk-free lending – this is taken to reflect the rate at which individuals are willing to exchange present for future consumption. Taking this as the rate at which to discount in CBA is regarded as an application of consumer sovereignty in the intertemporal context.

If we take the welfare improvement test view of CBA, we could, as a practical matter, just base an estimate for r on observed behaviour in this way. Many proponents of this approach, however, argue that we should use equation 11.23 to derive a value for r from values for ρ, the utility discount rate, η, the elasticity of the marginal utility of consumption, and g, the growth rate. Then, some argue for a descriptive approach to the numbers and attempts have been made (see Pearce and Ulph, 1995, for references) to infer values for ρ and η from observed behaviour, which are then combined with historically based estimates for future g to calculate a value for r. Others take the view that a value for r should be calculated from ethically based values for ρ and η together with an informed view about prospects for g. There is some suggestion that, particularly in regard to ρ, the prescriptive position becomes more tenable the longer the lifetime for the project under consideration.

While there is less disagreement about discounting in CBA than was once the case, coming up with a number for the rate of discount is not a simple settled matter. As can be seen in Box 11.2, quite different numbers for r can be found coming from

Box 11.2 Discount rate choices in practice

What rate should an economist working for a government agency use for discounting in CBA? Here we consider the situation in two countries, the UK and the USA. In what follows here, we use the terminology and notation that we have used consistently in our discussions, rather than that of the sources cited here. Terms are used in varying ways in the literature, and this can lead to confusion and misunderstanding.

In the USA the Office of Management and Budget issued a requirement for all federal agencies of the executive arm of government to use 10%. In 1994, by Circular A.94 (which can be accessed at http://www.whitehouse.gov/omb/), this was changed to 7%. The circular considers using r together with the shadow pricing of capital, but rejects this in favour of δ, mainly on the grounds that shadow pricing is too difficult in practice. The 7% figure is an estimate of the pre-tax return on capital in the USA.

The US Environmental Protection Agency has produced *Guidelines for Preparing Economic Analyses* (as of early 2008, the version downloadable at http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/Guidelines.html/$file/Guidelines.pdf dates from 2000) for 'those performing or using economic analysis, including policy makers, the Agency's Program and Regional offices, and contractors providing reports to the EPA'. The guidelines make a distinction between intra- and inter-generational discounting. The dividing line between the two situations is not specified in years. Examples falling into the inter-generational category are given as 'global climate change, radioactive waste disposal, groundwater pollution, and biodiversity'. Everyday usage might suggest that a project lifetime of more than 30 years would qualify as inter-generational.

In regard to intra-generational projects the EPA guidelines advocate using r, and take a descriptive approach, arguing that this is what consumer sovereignty requires. It is argued that, other than in exceptional cases, no shadow pricing of capital is necessary, as the US economy is open to international capital flows. Based on an assessment of the post-tax return on risk-free lending, it recommends using a value for r in the range 2% to 3%. It also states that analyses should include presentation of undiscounted flows, and also include present value results following Office of Management and Budget directions, i.e. discounting at 7%.

Box 11.2 ***continued***

For inter-generational projects, the US EPA guidelines note the difficulties of a consumer sovereignty approach to a number for r where long time periods are involved, and introduce the relationship between ρ and r which is equation 11.23 here. They claim that the value for ρ is usually set at 0 on ethical grounds, and that the assumptions made about η and g in the literature then typically produce values for r in the range 0.5% to 3%. The recommendation is then that analyses of inter-generational projects should present results as for intra-generational projects, i.e. for rates in the range 2% to 7%, plus results for discounting at rates in the 0.5% to 3% range, and that discussion of this sensitivity analysis should 'include appropriate caveats regarding the state of the literature with respect to discounting for very long time horizons'.

In the UK the position on discounting provided in guidance from HM Treasury on the appraisal of public-sector projects has varied over time. In 1988 the 'test' discount rate was set at 5%. In 1991 this was raised to 6%. The most recent guidance is in the Treasury's 2003 edition of *The Green Book* (HM Treasury (2003) and at http://www.hm-treasury.gov.uk/economic_data_and_tools/greenbook/data_greenbook_index.cfm). *The Green Book* bases its position on a version of equation 11.23 in which the pure time preference and risk of extinction elements for ρ (discussed in Chapter 3 here) are considered separately. Its approach is descriptive – 'the evidence' is taken to suggest a value of 'around' 1.5% for ρ. The approach to η is also said to be based on evidence, which suggests a value of around 1. The growth rate g is taken to be 2%, based on a historical average of 2.1% for the UK over 1950 to 1998. With these values, r comes out at 3.5%.

There is no mention of shadow pricing of capital. Presumably this is because the UK is open to international capital flows. *The Green Book* states that for projects with lifetimes greater than 30 years, the discount rate should decline as follows:

Years ahead	31–75	76–125	126–200	201–300	301+
Discount rate	3.0%	2.5%	2.0%	1.5%	1.0%

The rationale for a declining long-term discount rate is given as 'uncertainty about the future'. No argument, or evidence, is cited for the particular numbers recommended.

The approach followed in *The Green Book* appears to have been influenced by a 1995 paper on the rate of discount for public sector appraisal in the UK, Pearce and Ulph (1995). Based on a mixture of prescriptive and descriptive considerations, Pearce and Ulph come up with a range of 0.9% to 5.0%, with a best estimate of 2.4% for the discount rate. They concluded that the Treasury rate in force at the time of their writing, 6%, was 'far too high'. Pearce and Ulph put the argument for shadow pricing capital, but do not actually recommend it. They do not, however, make the international capital mobility argument against shadow pricing. Pearce and Ulph do not distinguish between intra- and inter-generational discounting, nor do they advocate a discount rate declining with time for long-life projects.

The Stern Review (Stern 2006) on the economics of climate change, introduced in Box 3.1 here, paid a lot of attention to the rate of utility discount ρ, and did not look explicitly at the consumption discount rate r. However, an implied value of r can be obtained using the values given in the review, on a prescriptive basis, for ρ, η and g, which are 0.1%, 1, and 2%, in equation 11.23. On this basis, the review's value for r would be 2.1%. This is pretty much in the middle of the inter-generational rate range actually recommended by the US EPA, and very close to the UK Treasury recommendation for projects with lifetime 200 years. It is, however, lower than the rate used in some previous economic analyses of the climate change problem, and, as noted in Box 3.1, attracted some criticism on that basis – references are given in Box 3.1.

Leaving aside inter-generational projects, for the US EPA, and long lifetime projects, for the UK Treasury, we have here a range of recommended consumption discount rates from 7% to 3.5%. There is no reason for this rate to be equal across countries, of course – in terms of equation 11.23, countries may differ on one or all of ρ, η and g, on either a prescriptive or descriptive approach. Looking just at the UK, Pearce and Ulph have a best estimate of 2.4%, while the Treasury says 3.5%. Is this much of a difference? At 2.4%, £100 30 years hence has a present value of £49.09, whereas at 3.5% the present value would be £35.63. Clearly, it could matter a lot which of these rates is used. Clearly, sensitivity analysis, presenting results for different rates, will generally be appropriate. However, this may then leave a lot to be decided by the judgement of the decision maker for whom the CBA is being conducted.

what might be regarded as authoritative sources. And, not all economists now accept that r is the right thing to use anyway, and we now look briefly at an argument for an alternative approach to discounting public-sector project net benefits.

11.2.3.2 The social opportunity cost argument

Box 11.2 notes that in the USA the Office of Management and Budget, OMB, requires all federal agencies to discount at 7%, this being an estimate of the pre-tax return on capital in the USA. The OMB bases this requirement on the social opportunity cost of capital argument.

This argument starts with the observation that in the non-ideal world in which we actually live it is not true that $r = i = \delta$. One major reason for this is taxation. As already noted, if r is identified with the post tax return to risk-free lending then it is of the order of 3%. It is usual to identify δ with the marginal pre-tax rate of return on private investment. What is actually observed is average pre-tax rate of return. This is generally taken to be of the order of 5%, though as noted the OMB have 7% for the USA. The marginal rate of return would be higher than the average rate if private-sector investment did in fact consistently undertake higher-return projects before lower-return projects.

Given $\delta > r$, a project that passes the NPV test using r as discount rate may not pass the test if δ is used as the discount rate. The argument then is that public-sector project appraisal should use δ as discount rate, because otherwise public-sector projects will pass the test using r and use resources which had they been used in the private sector would have yielded a larger surplus. Public sector appraisal should, that is, use δ so as to properly measure the social opportunity cost of capital.

The first point to make about this argument is that it is confused. The point of discounting in CBA is to weight consumption gains and losses, net benefits, at different points in time so as to aggregate them. The proper way to do this is by using the consumption discount rate. The point that the social opportunity cost argument is making is about the real cost of the capital going into a public-sector project. The way to deal with this is by shadow pricing that capital, as previously discussed here, not by changing the discount rate.

The second point to be made is that, like the argument for shadow pricing capital, the social opportunity cost argument depends on the assumption that the public-sector project crowds out, is at the expense of, the marginal private-sector investment project. As we have seen, it is no longer considered appropriate to assume that crowding out takes place.

Despite its endorsement by the OMB, the social opportunity cost argument for a higher, than r, discount rate in public-sector project appraisal is wrong, and is now so regarded by most economists.

11.2.3.3 Social appraisal of private sector investment

Thus far we have been considering discounting in relation to public-sector investment projects, but we started our discussion of CBA by noting that it is also used to appraise government policies intended to affect private-sector behaviour, and private-sector investments with consequences that are externalities. Do these other contexts require any alteration to the conclusion that net benefits should be discounted at the consumption rate of discount? No. We explain this briefly by considering the appraisal of some government policy initiative that requires private-sector investment – a regulation prohibiting the release by manufacturing industry of some pollutant into the atmosphere, say.

The first question is whether or not the affected firms can pass on to their customers, in the form of higher prices, the costs of the additional investment required to comply with the regulation. If they can, then the costs of the regulation fall on consumers, and can be set against the gains to those individuals from the regulation, in the form of cleaner air, to give a stream of consumption changes, net benefits, to be discounted at the consumption discount rate. If they cannot, then the next question is whether or not this investment to meet the regulation displaces, or crowds out, other investment. If it did, the capital investment required to comply with the regulation would have to be shadow priced. But, as we have already noted, the answer to this question would now be seen as being negative – given the perfectly elastic supply of capital to an open economy there is zero crowding out, and shadow pricing is not called for.

11.2.3.4 Projects with long lifetimes

As can be seen from Table 11.8, discounted at the UK Treasury's prescribed rate of 3.5%, net benefits of $100 100 years out go into the NPV calculation as $3.21, and 200 years out as $1.03. If we consider nuclear power, for example, this means that the future costs of waste storage would count little in deciding now on whether to build a nuclear power plant. As noted in Box 11.2, both the UK Treasury and the US EPA require the use of lower discount rates for the appraisal of projects with long lifetimes. In the former case, 'long' is explicitly stated to be more than 30 years, while the latter refers to projects with 'inter-generational' consequences.

The UK Treasury states that for it the 'main rationale' for its requirement is 'uncertainty about the future' (page 98 of *The Green Book*), citing the work of Weitzman (1998). His argument is, roughly, that the rate of return to capital in the distant future is highly uncertain, and that it makes sense to proceed on the basis that it is the lowest of the conceivable numbers. The uncertainty and the spread of the conceivable numbers increase with the distance in the future.

In fact it is not necessary to appeal to uncertainty to make a case for having the discount rate fall as futurity increases. Such a case can be made on the basis of the descriptive approach to the utility and/or the consumption rate of discount. There is now evidence, from surveys and laboratory experiments, that people discount the near future at a higher rate than the distant future. This is often referred to as 'hyperbolic' discounting. In exponential discounting, which we have been considering thus far, the discount rate is a constant so that the discount factor declines exponentially – see Figure 3.7 for a plot of utility weights, or utility discount factors, against time. A formal model of hyperbolic discounting has the discount rate itself declining at a constant rate with time.

As summarised in Heal (2005), the empirical evidence suggests that for periods up to five years, people use discount rates higher than those now recommended for CBA – 15% and upwards. For 10 years this drops to around 10%, 5% for 30 to 50 years, and 2% for 100 years. While these numbers are consistent with a stepwise version of hyperbolic discounting, it should be noted that, for less than 50 years, they are higher than the UK Treasury required rate.

The US EPA mainly bases its argument for a consumption discount rate of 0.5% to 3% for intergenerational products on the derivation of that rate from the utility rate, the elasticity of the marginal utility of consumption and the growth rate, as in equation 11.23. No details are provided as to the values it considers appropriate for the inputs on the right-hand side of the equation.

While there is an increasing spread of agreement that projects with long lifetimes should be discounted at different, lower, rates, there is not a lot of agreement about exactly what the basis for that is, nor about the numbers that should be used. The fact has to be faced that the whole business of discounting costs and benefits remains somewhat contentious, and that the importance of disagreements to appraisal outcomes increases with project lifetime. Many projects with environmental impacts are properly considered projects with long lifetimes. In what follows we shall assume exponential discounting at a constant, and typically un-specified, rate.

11.3 Cost–benefit analysis and the environment

We now wish to discuss CBA in relation to the environment. This is a wide field with an extensive literature. In order to fix ideas we will consider a wilderness forest area, in which some development – a mine, a hydroelectric plant, timber harvesting or perhaps a theme park and tourist resort – is proposed. Currently, the area is relatively inaccessible and is used only for low-intensity recreation, such as backpacking for example. It also provides habitat for numerous species of flora and fauna, and thus plays a role in biodiversity conservation. If the development goes ahead, the area's value to wilderness recreationalists will be reduced, as will its effectiveness in biodiversity conservation. The question at issue is whether the development project should be allowed to go ahead.

For economists, this question is to be answered by CBA. The development project should be appraised by the methods discussed in the previous section of this chapter, taking due account of any losses suffered by individuals on account of the reduction in its wilderness recreation and conservation services. These services do not pass through markets, so they cannot be taken into account by a project appraisal which calculates NPV using market prices. Economists have developed a variety of techniques for 'non-market valuation' so that services such as wilderness recreation and biodiversity conservation can be included in CBA. We consider these techniques in some detail in the next chapter. For now, we shall simply say that the essential point of these techniques is that the intention is to ascertain what the affected individuals collectively would be willing to pay if there were markets for these services. This is what the logic of bringing them within the ambit of applied welfare economics requires.

To emphasise that, in circumstances where a project involves environmental impacts that are not valued in markets, a proper CBA should take account of such impacts, let us call it environmental cost–benefit analysis, ECBA. The scope of ECBA is much wider than the appraisal of development projects in wilderness areas, but looking at it in that context does bring out the most important issues.

11.3.1 Environmental cost–benefit analysis

We know that to do a cost–benefit analysis we calculate

$$\text{NPV} = \sum_{t=0}^{t=T} \frac{\text{NB}_t}{(1+r)^t}$$

and that the project should go ahead if NPV > 0. Net benefits are the excess of benefits over costs in each period and we can write

$$\text{NPV} = \sum_{t=0}^{t=T} \frac{B_t - C_t}{(1+r)^t}$$

with B for benefits and C for costs. In ECBA benefits and costs are to include, respectively, the value of environmental improvement and of environmental deterioration consequent upon going ahead with the project. In fact, in discussing ECBA it is convenient for expositional purposes to keep ordinary benefits and costs separate from environmental benefits and costs. By 'ordinary' benefits and costs we mean the value of standard, non-environmental, outputs from and inputs to the project – such as, in the case of a mine, the extracted ore on the benefit side, and on the cost side inputs of labour, capital equipment, fuel and so on. As noted in the previous section, ideally all these inputs and outputs would be expressed in consumption-equivalent terms.

Let B_d be the discounted value of the ordinary benefit stream over the project lifetime, and let C_d represent the discounted value of the ordinary cost stream over the project lifetime, so that ignoring environmental impacts we can write:

$$\begin{aligned}\text{NPV} &= \sum_{t=0}^{t=T} \frac{B_t - C_t}{(1+r)^t} = \sum_{0}^{T} \frac{B_t}{(1+r)^t} + \sum_{0}^{T} \frac{C_t}{(1+r)^t} \\ &= B_d - C_d\end{aligned}$$

To denote an NPV that ignores environmental impacts, use NPV′ for it. Then, taking account of environmental impacts, the 'proper' NPV is given by

$$\text{NPV} = B_d - C_d - \text{EC} = \text{NPV}' - \text{EC} \tag{11.27}$$

where EC is the present value of the stream of the net value of the project's environmental impacts over the project's lifetime. Note that in principle EC could be negative, with the value of environmental benefits exceeding the value of environmental costs, so that NPV > NPV′. The net value of the environmental consequences of the project, could, that is, be such as to strengthen, rather than weaken, the case for the project. However, we shall assume that EC is positive. In fact, it will be convenient to make the stronger assumption that there are no desirable environmental consequences of going ahead with the project, that it causes only environmental damage. This assumption appears to sit well with development in a wilderness area, and is what is typically assumed about such development in the literature. Given this, EC stands for 'environmental cost'. It could also be taken to stand for 'external cost' as the unpriced environmental damages are externalities associated with the project.

Using equation 11.27 the ECBA decision rule is that the project should go ahead if

$$NPV' = B_d - C_d > EC \tag{11.28}$$

The application of this criterion requires the identification and measurement of the impacts on the wilderness area, and then their valuation and aggregation to arrive at EC, which is a monetary measure of the environmental benefits of not going ahead with the project.

Assuming that the environmental impacts on individuals can be identified and measured, the basic strategy for valuation is to treat them as arguments in utility functions, to treat them, that is, in the same way as ordinary produced goods and services. Then, as discussed in the next chapter, demand theory can be used to establish the existence and nature of monetary measures of the impacts. The implementation of this ECBA approach to social decision making then requires the estimation of the sizes of the appropriate monetary measures for affected individuals and their aggregation to obtain an estimate for EC.

Now clearly, if NPV′ < 0 then the project should not go ahead, independent of any consideration of the environmental damage that it might entail. A development of this observation, where NPV′ has been assessed as some positive number, is to ask: how large would EC have to be in order, according to ECBA, for the project not to go ahead? The answer is obvious. The project should not go ahead if

$$EC \geq NPV' = B_d - C_d$$

so that

$$EC^* = NPV' = B_d - C_d \tag{11.29}$$

defines a threshold value for EC. For EC ≥ EC* the project should not go ahead.

This suggests that what we can call an 'inverse ECBA' might usefully precede or accompany an ECBA. ECBA itself requires the identification, measurement and valuation of the project's environmental impacts on affected households. Such an exercise involves non-trivial expenditures, and may, nevertheless, produce results that do not command universal assent, as discussed in the next chapter. Inverse ECBA simply means properly figuring NPV′, and then asking what average valuation of the environmental impacts would have to be to produce a negative verdict on the project. It involves, that is, calculating the threshold for total environmental cost, EC*, and dividing it by *N*, the size of the relevant population of individuals. In some cases the result of this calculation will be such a small amount that it could be generally agreed, or at least widely agreed, that the project obviously should not go ahead.

Even where this is not the case, and a serious attempt to estimate EC/*N* is undertaken, the value for EC*/*N* will provide a useful benchmark against which to consider the result for EC/*N* produced by the application of the techniques to be considered in the next chapter. Given the problems that will be seen to attend the results from the various environmental valuation techniques, estimating EC/*N* as, say, 10 times EC*/*N* produces a very different decision situation from estimating EC/*N* as, say, 1.5 times EC*/*N*. In the former case one might be reasonably confident that the project should not go ahead; in the latter case much less so.

Thinking about wilderness development projects in inverse ECBA terms directs attention to the question of the size of *N*, the number of individuals that would be affected if the project went ahead. In regard to recreational use of the undeveloped area, this would be the number of visitors, which would be of the order of tens of thousands perhaps. In regard to biodiversity conservation, it is not necessary for an individual to actually, or even potentially, be a visitor to the area for them to be affected by a reduction in its conservation value. There is evidence from a variety of sources that many individuals are willing to pay to promote wildlife conservation in areas that they will never visit. In regard to conservation, the whole population of the nation in which the threatened wilderness area is located is generally seen as the relevant population, in which case we are looking at a value for *N* of the order of millions. Indeed, for wilderness areas that are internationally famous for their wildlife it could plausibly be argued that it is a proportion (the relatively affluent inhabitants of the developed world) of the global population that is the relevant population, making *N* of the order of tens or hundreds of millions. In that case, the per capita valuation of the conservation cost of development required to give a value for EC greater than the project's NPV′ may be very small.

Box 11.3 illustrates some of these points about inverse ECBA for an Australian project appraisal.

Box 11.3 Mining at Coronation Hill?

In 1990 there emerged a proposal to develop a mine at Coronation Hill in the Kakadu national park, which is listed as a World Heritage Area. The Australian federal government referred the matter to a recently established advisory body, the Resource Assessment Commission, which undertook a very thorough exercise in environmental valuation using the Contingent Valuation Method (to be considered in the next chapter), implemented via a survey of a sample of the whole Australian population. This exercise produced a range of estimates for the median willingness to pay, WTP, to preserve Coronation Hill from the proposed development, the smallest of which was \$53 per year. In CBA, WTP to preserve from the project is taken as the measure of the environmental damage value consequent on going ahead with the project. If it is assumed, conservatively, that the \$53 figure is WTP per household, and this annual environmental damage cost is converted to a present value capital sum in the same way as the commercial NPV for the mine was calculated, the EC to be compared with the mine NPV is, in round numbers, \$1500 million. This 'back of the envelope' calculation assumes 4 million Australian households, and a discount rate of 7.5%.

The publication of this result gave rise to much comment, mainly critical, and some hilarity. It was pointed out that given the small size of the actual area directly affected, the implied per hectare value of Coronation Hill greatly exceeded real estate prices in Manhattan, whereas it was 'clapped-out buffalo country' of little recreational or biological value. In fact, leaving aside environmental considerations and proceeding on a purely commercial basis gave the NPV′ for the mine as \$80 million, so that the threshold per Australian household WTP required to reject the mining project was, in round numbers, \$3 per year, less than one-tenth of the low end of the range of estimated household WTP on the part of Australians. Given that Kakadu is internationally famous for its geological formations, biodiversity and indigenous culture, a case could be made for extending the existence value relevant population, at least, to North America and Europe. On that basis, the size of WTP per Australian household required to block the project would be much smaller than \$3.

In the event, the Australian federal government did not allow the mining project to go ahead. It is not clear that the CVM exercise actually played any part in that decision. What is clear is that even if the CVM result overestimated true Australian WTP by a factor of 10, it would still be the case that ECBA would reject the mining project even if the Australian population was taken to be the entire relevant population.

Source: Resource Assessment Commission (1991).

11.3.2 The Krutilla–Fisher model

NPV is the result of discounting and summing over the project's lifetime an annual net benefit stream which is

$$NB_t = B_{d,t} - C_{d,t} - EC_t \qquad (11.30)$$

where $B_{d,t}$, $C_{d,t}$ and EC_t are the annual, undiscounted, amounts for $t = 1, 2, \dots, T$, and where T is the project lifetime, corresponding to the present values B_d, C_d and EC. The environmental costs of going ahead with the project, the EC_t, are at the same time the environmental benefits of not proceeding with it. Instead of EC_t we could write $B(\mathrm{P})_t$ for the stream of environmental benefits of preservation.[4] If we also use $B(\mathrm{D})_t$ and $C(\mathrm{D})_t$ for the benefit and cost streams associated with development when environmental impacts are ignored, so that $B(\mathrm{D})_t - C(\mathrm{D})_t$ is what gets discounted to give NPV′, then equation 11.30 can also be written as:

$$NB_t = B(\mathrm{D})_t - C(\mathrm{D})_t - B(\mathrm{P})_t \qquad (11.31)$$

It will now be convenient to treat time as continuous, so that instead of

[4] In some of the literature on wilderness development there would also be distinguished $C(P)$ for the costs of preservation, where such costs are those associated with, for example, managing the national park set up to realise preservation. Here we do not explicitly introduce such costs as this simplifies without any essential loss. Our $B(P)$ can be interpreted as preservation benefits net of any such costs. Clearly, such an interpretation does not substantially affect the plausibility of the assumptions about relative price movements to be introduced shortly.

$$\text{NPV} = \sum_{0}^{T} \{B(\text{D})_t - C(\text{D})_t - B(\text{P})_t\}/(1 + r)^t$$

we use

$$\text{NPV} = \int_0^T \{B(\text{D})_t - C(\text{D})_t - B(\text{P})_t\} e^{-rt} \text{d}t$$

which can be written:

$$\text{NPV} = \int_0^T \{B(\text{D})_t - C(\text{D})_t\} e^{-rt} \text{d}t - \int_0^T B(\text{P})_t e^{-rt} \text{d}t \tag{11.32}$$

Krutilla and Fisher (1975) introduced important and persuasive arguments as to why it should be assumed that the value of wilderness amenity services will, relative to the prices of the inputs to and outputs from development, be increasing over time. The arguments concern substitution possibilities, technical progress and the income elasticity of demand for wilderness services.

In the Krutilla–Fisher model the development option is seen as producing extracted intermediate outputs. It is typically the case that these intermediate outputs have relatively close substitutes. Moreover, the degree of substitutability tends to increase over time as technical knowledge develops. If we consider hydroelectric power, for example, it is clear that this form of power has many close substitutes, such as power from fossil fuel and nuclear sources. Technological advances have increased these substitution possibilities in recent decades, and will almost certainly continue to do so in the foreseeable future. If fusion power were to become technically and commercially viable, very long-term substitution possibilities will have been opened up. Finally, one would expect that rising demand for the extractive outputs of the development can be met at decreasing real costs over time, as energy production and conversion benefits from technological innovation.

This contrasts strongly with the case of wilderness preservation benefits. The substitution possibilities here are often effectively zero, and there is no reason to suppose that they will become greater due to technical progress. Second, it is plausible and consistent with the evidence that environmental amenity services, and especially those of wilderness areas, have a high income elasticity of demand. But, third, technological progress itself cannot augment the supply of such services.

With economic growth and technological change it is reasonable to assume a tendency for the relative value of amenity services from undeveloped environmental assets to increase. A simple way to introduce this into equation 11.32 is to assume that preservation benefits grow at the rate a, while development benefits and costs are constant, so that

$$\text{NPV} = \int_0^T \{B - C\} e^{-rt} \, \text{d}t - \int_0^T \{P\text{e}^{at}\} \text{e}^{-rt} \text{d}t \tag{11.33}$$

where B and C are the constant development benefit and cost flows, while $P\text{e}^{at}$ is the growing flow of preservation benefits. This can be written as

$$\text{NPV} = \text{NPV}' - \int_0^T P\text{e}^{-(r-a)t} \text{d}t \tag{11.34}$$

Note here, first, that for $a > 0$, NPV will be less than for $a = 0$, for given NPV′. This means that for a given NPV′, a development project is less likely to pass the intertemporal allocative efficiency test if the Krutilla–Fisher arguments are accepted and incorporated into ECBA. The second point to note is that if $a = r$, then, in effect, preservation benefits are not discounted. If it were to be assumed that $a > r$, then those benefits would effectively get discounted at a negative rate, and the discounted stream for P_t would itself be growing over time.

Now, let us suppose that $T \to \infty$. There are two reasons for making this assumption. First, it means that we can use a standard mathematical result which greatly simplifies the analysis.[5] The result is that

$$\int_0^\infty x\text{e}^{-rt} = x \int_0^\infty \text{e}^{-rt} = \frac{x}{r}$$

[5] This result is proved in, for example, Chiang (1984): see p. 464. The equivalent result in a discrete-time context is established in chapter 8 of Common (1996).

where x is some constant. The present value of a constant sum x for ever is x divided by the relevant interest rate r. This result is actually quite a good approximation where T is of the order of 100. For $r = 0.05$, the present value of

x for 50 years is $0.9128(x/r)$
x for 75 years is $0.9742(x/r)$
x for 100 years is $0.9924(x/r)$
x for 125 years is $0.9978(x/r)$

and for T fixed, the approximation gets closer as r increases.

The second reason for having $T \to \infty$ is that in practice for wilderness development projects, T is appropriately taken to be a very large number. T is the project lifetime, which is defined not by the date at which the project ceases to serve the function for which it was undertaken, but the date at which the longest-lived consequence of the project ceases. Thus, for example, if the project is a mine with an extraction life of 50 years, but where vegetation will take 200 years to recover after the closure of the mine, then T is 250.

Applying the result above in equation 11.34, it becomes:

$$\text{NPV} = \text{NPV}' - P/(r - a) \tag{11.35}$$

Note that as a increases, so $P/(r - a)$ increases, so that for NPV′ given, NPV decreases. This is illustrated in Table 11.9, which shows how the second term in equation 11.35 varies with the value of a for $r = 0.05$ and $r = 0.075$, where $P = 1$. Note that the long-term rate of economic growth is generally taken to be around 2.5%, that is, 0.025, and that it can be argued that this provides a plausible lower bound for the value that should be assumed for a. Note also that for $a > r$ the standard result used to go from equation 11.34 to equation 11.35 does not hold, because, as noted above, the discounted P_t are growing over time. For P at some value other than 1, the entries in Table 11.9 are the factors by which P, the current value of preservation benefits, would be multiplied to give the value of their loss for ever.

Table 11.9 $P/(r - a)$ for $P = 1$

a	For $r = 0.05$	For $r = 0.075$
0	20	13.33
0.01	25	15.37
0.02	33.33	18.18
0.03	50	22.22
0.04	100	28.57
0.05	∞	40
0.06		66.67
0.075		∞

11.3.3 Discount rate adjustment?

Conservationists sometimes argue that when doing an ECBA of a project giving rise to long-lasting environmental damage, a lower discount rate should be used as this will give more weight to environmental costs far into the future, thus making it less likely that the project will get the go-ahead. As we saw in section 11.2.2, many (but not all) economists would agree that projects with long lifetimes should be appraised using a lower discount rate than used for projects with comparatively short lifetimes. However, it is not always true that reducing the discount rate for a project with long-lasting environmental damage will work to shift the appraisal in the direction of rejection.

We can consider what is involved here by first writing equation 11.33 as

$$\text{NPV} = \{B - C\}\int_0^T e^{-rt}\,dt - P\int_0^T e^{-(r-a)t}\,dt$$

or, using D for net development benefits,

$$\text{NPV} = D\int_0^T e^{-rt}\,dt - P\int_0^T e^{-(r-a)t}\,dt$$

Using the standard result from above as an approximation for very large T, this is:

$$\text{NPV} = \frac{D}{r} - \frac{P}{r - a} \tag{11.36}$$

Now, thus far in treating $D = (B - C)$ as constant over all t we have overlooked one feature of development projects, which is that they typically involve a short initial period with capital expenditure but no sales revenue – digging the mine or building the dam for the hydroelectric facility – followed by a long period with running costs and sales revenues.

The stylised facts here can be captured by rewriting equation 11.36 as

$$\text{NPV} = -X + \frac{D}{r} - \frac{P}{r-a} \tag{11.37}$$

where X is the initial start-up cost, which does not get discounted.

Suppose that X is 1000, D is 75 and P is 12.5 in monetary units, say millions of pounds. Consider first a case where it is assumed that $a = 0$. Then

$$\text{NPV} = -X + \frac{D-P}{r} \tag{11.38}$$

and for $r = 0.055$ NPV is 136.37, while for $r = 0.045$ NPV is 388.89. For these numbers, lowering the discount rate has increased the NPV. This is because reducing the discount rate affects both development net benefits and environmental costs in the same way. From Equation 11.38 it is clear that, for $(D - P)$ positive, reducing r will increase NPV. Of course, to the extent that both D and P are not everlasting, we are dealing here with an approximation. But for time horizons of 100 years or more, it will be a close approximation.

Now suppose that it is assumed that a in equation 11.37 is 0.025. In this case, for $r = 0.055$ the NPV is –53.03, while for $r = 0.045$ the NPV is 41.67. Reducing the discount rate shifts the ECBA decision from rejection of the project to going ahead with it. Lowering r increases D/r by more than it increases $P/(r - a)$. The point here is not that reducing the interest rate for this kind of project will increase the NPV for any values for D and P and any initial r. From equation 11.37 it is clear that this would not be the case. The point is to provide an illustration of a counter-example to the proposition that reducing r will always work against projects with damaging and long-lasting environmental effects. That proposition is not generally true. While reducing r gives more weight to environmental damage very far into the future, it also gives more weight to net development benefits moderately far into the future, and far into the future if they continue that long.[6]

11.3.4 Objections to environmental cost–benefit analysis

In order to do ECBA it is necessary to figure out what EC is. The non-market valuation methods by which economists seek to measure EC are considered in the next chapter. As we shall see there, there is some dispute about the accuracy of the methods. Some argue that the methods do not produce reliable information for inclusion in ECBA. Some, mainly economists, who take this position consider that the existing methods can be improved so as to provide reliable information, and/or that new methods can be developed that will produce reliable information. Others take the view that there are inherent limitations to the accuracy of non-market valuation, and hence to that of ECBA. As we shall see in the next chapter, the environmental valuation methods require that environmental impacts are arguments in well-behaved utility functions. Some, economists and others, argue, and provide evidence to support the argument, that this assumption is not satisfied, in that people do not, in fact, generally relate to the environment in this way. If this is true, then non-market valuation methods cannot do what ECBA requires them to do.

These arguments will be reviewed in the next chapter, after we have worked through the methods to which they relate. Here we are concerned with a different sort of objection to ECBA. Many people, who are mainly but not exclusively non-economists, take the view that it is simply the wrong way, on ethical grounds, to inform social decision making where there are serious environmental impacts at issue.

ECBA is applied welfare economics. We discussed the ethical basis for welfare economics in Chapter 3. Here we can summarise by saying that welfare economics is based on a particular form of utilitarianism, which is 'consequentialist' and 'subjectivist' in nature. It is consequentialist in that actions are to be judged in terms of their consequences for human individuals. It is only human individuals that are of interest – only humans have 'moral standing'.

[6] The analysis here is based on Porter (1982), where there is a more rigorous and extended discussion. See also chapter 8 of Common (1995) for a detailed numerical illustration of these points.

It is subjectivist in that the measure of what is good for a human individual is that human individual's own assessment. The individual's assessment is to be ascertained from his or her preferences as revealed in behaviour. All of this is roughly encapsulated in the idea of 'consumer sovereignty'. There are two classes of ethical objection to this way of proceeding.

The first accepts that only human individuals have moral standing but rejects consumer sovereignty, arguing that individual preferences are a poor guide to individual human interests. Following Penz (1986), four particular arguments can be distinguished:

1. Individuals may be inadequately informed as to the consequences for themselves of the alternatives they face.
2. Individuals may be insufficiently deliberative in assessing the consequences of alternative choices.
3. Individuals may lack self-knowledge in the sense that they cannot properly relate the consequences of alternative choices to their preferences.
4. Individuals' preferences may not reflect their true interests due to 'preference shaping' arising from socialisation processes and advertising.

These arguments are not restricted to the environmental context, but have been argued to have special force there: see, for example, Vatn and Bromley (1995) and Norton (1994). The philosopher Mark Sagoff (1988, 1994, 1998) particularly has argued against social choice on the basis of 'preference satisfaction', and for social choice by 'deliberative citizens' rather than 'consumers' in the environmental context. His point is that where serious environmental issues are involved, it is simply wrong to appeal to the self-interested preferences that might be acceptable as the criterion for deciding how much whisky as opposed to beer to produce. Sagoff argues that the correct way to make decisions with serious environmental implications is as the result of the deliberations of citizens – individuals whose views reflect their assessment of what is good for society.[7]

A second class of argument is that the scope of ethical concern should not be restricted to humans, that animals and plants (and in some versions non-living entities) should have 'moral standing': see, for examples, Naess (1972), Goodpaster (1978), Regan (1981) and Singer (1979, 1993). Booth (1994) argues that 'cost–benefit analysis cannot be legitimately applied where, as they should be, non-human natural entities are viewed as morally considerable' (p. 241), and that the ethically correct principle for social decision making is that 'Destruction of the natural environment shall not be undertaken unless absolutely necessary to maintain the real incomes of all human individuals at a level required for the living of a decent human life' (p. 251). This has affinities with the safe minimum standard idea, to be discussed in Chapter 13. That idea is based upon a consequentialist theory restricted to human interests, but recognises the uncertainties that attend predicting the future costs of current environmental damage.

11.3.4.1 Sustainability and environmental valuation

We considered sustainability in Chapters 2 and 3, where we argued that a commitment to sustainable development involves an appreciation of the facts of economy–environment interdependence and an ethical position. We saw that such a commitment could take the form of adopting a different objective function from the one routinely used in welfare economics, or of retaining the standard objective function and maximising it subject to sustainability constraints.

Common and Perrings (1992) show that observing sustainability constraints may involve overriding the outcomes that are consistent with consumer sovereignty. Individuals' preferences may be consistent with the requirements for sustainability, but there is no guarantee that they will be, even if it is assumed that individuals are well informed. It follows that market failure correction, which is what ECBA and environmental valuation seek to deliver, is not sufficient for sustainability.

[7] It should be noted that self-interest as assumed in economics does not exclude the possibility of altruism – other individuals' consumption could well be arguments in my utility function with positive derivatives (negative derivatives would imply envy). Sen (1977) distinguishes between this kind of altruism, which he calls 'sympathy' and altruism as 'commitment' which is where my concern for others is based on ethical principles and could involve my acting in their interests even though it reduces my own utility. Commitment would be a characteristic of Sagoff's 'citizens' but not of his 'consumers'.

Suppose that we could ascertain accurately the aggregate monetary measure of the loss that consumers would suffer as the result of a decline in some environmental indicator. It does not follow that an ECBA on the project involved would produce an outcome consistent with sustainability requirements. It may be, for example, that the project would lead to the extinction of some species of termite that plays a key role in ecosystem function, and hence loss of resilience, but that EC would be, nonetheless, insufficient to stop the project. There is a reason for the choice of this example. Ecologists understand that termites do, in fact, play key roles in ecosystem function. There are good reasons – introspection and evidence from non-market valuation exercises – to suppose that the monetary measure of the loss suffered on account of the extinction of a termite species would be small.

In Chapter 3 we noted that 'weak' and 'strong' sustainability are different views about substitution possibilities rather than different views about what sustainability is. Strong sustainability proponents argue for the maintenance of natural capital on the grounds that human-made capital cannot substitute for it so as to permit constant consumption. Weak sustainability proponents, a group that includes most economists, argue for keeping the total stock of capital, human-made and natural, intact, and consumption constant, by substituting human-made for natural capital as the latter is depleted. This is not an ethical difference. Weak and strong sustainabilists have the same concern for intergenerational justice. They differ about the facts, the circumstances in which what that concern means in terms of action must be worked out.

Some of those who object to ECBA do so on the grounds that it implicitly involves the same assumptions about substitution possibilities as the weak sustainability position does, which assumptions are, in fact, incorrect. As noted above, there is no reason why a properly conducted ECBA would not allow a project known to entail species extinction to go ahead. The critics argue that this means that it is effectively being assumed that the services that the species provides can be substituted for by some other species and/or human-made capital, and that this assumption is wrong. They would argue that the domain of ECBA should be limited to cases where it is known that the project in question will not have impacts that entail the loss of environmental services for which there is no substitute. Given that these critics generally assume that possibilities for substituting for environmental services are very limited, this argument would greatly limit the range of applicability of ECBA.

11.3.5 Alternatives to environmental cost–benefit analysis

In order to briefly review the nature of some of the alternatives to ECBA that have been advocated, it will be convenient to consider a simple constructed example of a decision-making problem.[8]

Suppose that there are two towns linked by a four-lane highway built before both grew rapidly in population. The highway is frequently affected by severe traffic jams, and the government is considering three options for dealing with this problem. Option A is simply to build another four-lane highway between the two towns. Option B is to do that but to reserve one lane in each direction for specially built buses, with a view to reducing the emissions of CO_2 per person-mile travelled on this route. The third option considered is to build a new railway link rather than a new highway. It is thought that this could further reduce emissions and have less impact on wildlife and visual amenity.

However the decision is eventually to be taken, the first step is to assemble the basic information about each option in terms of costs, impact on the perceived problem, and environmental impact. This involves having engineers produce designs and hence costings for each option. Given the designs, the engineers can estimate the impact of each option on traffic flows on the existing highway and the new facilities, and hence the impact on the congestion problem. Traffic flow data will also permit of estimates of the CO_2 emissions associated with each option. Given the designs and routes, environmental scientists can be asked to assess the impacts on wildlife and landscape amenity values.

[8] The example used is based on one provided in Janssen and Munda (1999) who go into more detail on multi-criteria analysis.

This whole exercise would often be referred to as an environmental impact assessment, or an environmental and social impact assessment, or an impact assessment. The point is that at this stage what is at issue is just what would happen under each option – the objective here is not to evaluate the options, but simply to determine what they each involve. The complete separation of impact assessment from evaluation is possible conceptually, and helps to make clear what is involved in the various approaches to the evaluation and decision-making stages. In practice the separation is not clear-cut. Some commentators use the term 'environmental impact assessment' to include processes which are actually about evaluation rather than impact description. Many accounts of cost–benefit analysis focus exclusively on the evaluation of impacts, which can suggest that assessing the impacts is less of a problem than their evaluation. In fact, for many projects impact assessment is itself a large and difficult task.

It also needs to be noted that it would really be more descriptive of what is involved at this stage to refer to impact estimation rather than assessment. What will happen in the future under each option cannot be known with certainty in any of the dimensions of impact. Even cost data are estimates, which in the case of large engineering projects almost always turn out in the event to be significant underestimates. In what follows here we largely ignore imperfect knowledge of project impacts; Chapter 13 is mainly about the implications of risk and uncertainty for the ECBA approach to decision making.

One way of thinking about the distinction between impact assessment and evaluation is in terms of the role of expertise. Impact assessment is that part of the overall decision-making process where most people would regard it as appropriate to rely primarily on expert opinion. If we want an estimate of the effects on CO_2 emissions of alternative transport projects, for example, it would generally be agreed that it is better to ask trained traffic engineers and economists than to conduct a poll of a random sample of the population. If, on the other hand, it is a matter of choosing between two projects with given impacts, it could be argued that the choice should not be left to experts, but should reflect the preferences of the affected population as between the two sets of impacts. Impact assessment is that part of project appraisal that it would be generally agreed should be left to the relevant experts.

For our illustrative transport problem, assume that the impact assessment has been done and produces the data shown in Table 11.10. The impacts of the three options on the problem that is the origin of the various options, traffic jams and extended journey times, are measured in terms of (estimated) millions of hours saved per year. In terms of time savings, the more costly the option the less effective it is. CO_2 emissions effects are measured as tonnes arising per year under each option, and incurring more cost does more to reduce emissions. Whereas cost, time savings and emissions estimates can all be expressed quantitatively, we are assuming that the expert assessment of the wildlife and amenity impacts can only be expressed qualitatively. This is what is actually the case for many types of impact considered in actual environmental impact assessment exercises.

Before looking at some of the alternatives to it, let us briefly consider how ECBA would be used to appraise these projects and make a decision as between them. In our earlier account of project appraisal and cost–benefit analysis, we assumed that a decision had to be made as to whether or not to go ahead with a single project. Whether looked at from the commercial or the social perspective, the rule is to go ahead with the project if it has positive net present value – what differs as between commercial and social appraisals is that the latter takes account of costs and benefits that do not have market prices attached to them. It will be clear from our account of the logic of the net present value criterion, and from the earlier discussion of intertemporal efficiency,

Table 11.10 Options for reducing traffic delays

	A. Highway	B. Highway and Buses	C. Railway
Cost 10^6£	250	300	500
Time Saving 10^6 hours per year	10 000	8000	6000
CO_2 Emissions 10^3 tonnes per year	1 000	800	200
Wildlife and Amenity Qualitative	Bad	Bad	Moderate

that where a choice has to be made between two projects, the one with the higher NPV should be undertaken. Where there are several competing projects they should be ranked by NPV.

In the case now being considered, the ECBA decision rule is to adopt the option which has the highest NPV, provided that that is positive. Implementing this rule means setting out the time profile of each option in terms of arising flows of costs and non-monetary impacts, assigning monetary values to those impacts, and then using the agreed discount rate to arrive at an NPV figure for each option. The means by which monetary values would be assigned are discussed in detail in the next chapter. Broadly, time savings could be valued based on observed data about earnings per unit time, whereas there are no observed data that could be used to put a money value on either CO_2 emissions or wildlife and amenity impacts – people would have to be asked about their willingness to pay to secure improvements or avoid deteriorations here.

The important point is that it is the preferences of those affected that are to be used to evaluate the options. As already noted, objections take two basic forms. One argues that those preferences cannot be ascertained accurately. The other is that preferences are the wrong way to evaluate the options.

We now consider some alternatives to ECBA in the context of this simple constructed example.

11.3.5.1 Cost-effectiveness analysis

The basic idea of cost-effectiveness analysis is to select the option which achieves some specified objective at the least cost. Suppose, for example, that it had been decided that the minimum acceptable time saving was 8000 million hours per year. In that case the impact assessment rules out Option C, the railway. If just the monetary costs of construction are considered, Option A would be selected as it (over)achieves the target and costs less than the other option that meets the objective. However, there is no reason in principle why the monetary valuation methods that would be used to conduct an ECBA could not be used to bring emissions and wildlife and amenity impact into the ambit of the costs considered. In that case, people's preferences could be such that the poorer performance of A in emissions terms would lead to B being the selected option. Using preferences in this way would, of course, expose cost-effectiveness analysis to the same criticism as ECBA. In any case, cost-effectiveness involves, as ECBA does not, giving absolute priority to one aspect of performance.

11.3.5.2 Multi-criteria analysis

Multi-criteria analysis, MCA, is usually described as analysis which uses the preferences of decision makers to resolve situations where, as in our constructed example, the options get ranked differently on the various criteria that are considered relevant. We will come back to the matter of using the preferences of the decision makers after explaining how MCA uses preferences. There are actually many MCA methods, differentiated by the way in which the option evaluations on different criteria are combined to produce a single choice of option. Here we will use the simplest method, weighted summation, to illustrate what MCA basically involves: more detailed accounts of MCA and the methods that it can utilise are provided in the references given in the Further Reading section at the end of the chapter.

Some of the methods that have been proposed for MCA can work with qualitative data, but the weighted sums method cannot, so the first step is to convert the qualitative data in Table 11.10 to quantitative data. This is done by simply identifying each qualitative rating with a number as in

Bad	Moderate	Slight
3	2	1

in which case the evaluations of the three options on the four criteria are:

	A. Highway	B. Highway and Buses	C. Railway
Cost 10^6£	250	300	500
Time Saving 10^6 hours per year	10 000	8000	6000
CO_2 Emissions 10^3 tonnes per year	1 000	800	200
Wildlife and Amenity Qualitative	3	3	2

The numbers for the Wildlife and Amenity assessments could, of course, have been assigned the other way round, with a higher number going with a smaller impact. One of the Problems at the end of the chapter invites you to see the effect that that would have on what follows here.

The next step is to convert the data to dimensionless form, so as to permit aggregation. This is done by expressing the criterion outcome for each option as a ratio to the best outcome for the criterion which is set equal to 1. Consider Time Saving. On this criterion Option A is the best, so it gets set at 1, B at 8000/10 000 = 0.8, and C at 6000/10 000 = 0.6. For Cost, best is smallest, which is Option A at 250. Then B/A = 300/250 = 1.2 for which the reciprocal is 0.8333, and C/A = 500/250 = 2 with reciprocal 0.5. Proceeding in the same way for Emissions and Wildlife and Amenity, we get the dimensionless evaluation table:

	Highway	Highway and Buses	Railway
Cost	1.0000	0.8333	0.5000
Time Saving	1.0000	0.8000	0.6000
CO_2 Emissions	0.2000	0.2500	1.0000
Wildlife and Amenity	0.6667	0.6667	1.0000

The final step to is aggregate for each option across its criteria evaluations using weights that reflect the preferences of the decision makers, in terms of the relative importance attached to the various criteria. Suppose that the weights are agreed by the decision makers involved to be

Costs	0.3
Time Saving	0.3
CO_2 Emissions	0.2
Wildlife and Amenity	0.2

Then multiplying the entries in the dimensionless evaluation table by the weights and summing down columns gives

	Highway	Highway and Buses	Railway
Cost	0.3000	0.2500	0.1500
Time Saving	0.3000	0.2400	0.1800
CO_2 Emissions	0.0400	0.0500	0.2000
Wildlife and Amenity	0.1333	0.1333	0.2000
Sum	0.7733	0.6733	0.7300

so that the options are ranked: Highway, Railway, Highway and Buses.

For the weights

Costs	0.2
Time Saving	0.2
CO_2 Emissions	0.4
Wildlife and Amenity	0.2

we get

	Highway	Highway and Buses	Railway
Cost	0.2000	0.1667	0.1000
Time Saving	0.2000	0.1600	0.1200
CO_2 Emissions	0.0800	0.1000	0.4000
Wildlife and Amenity	0.1333	0.1333	0.2000
Sum	0.6133	0.5600	0.8200

and the ranking is: Railway, Highway, Highway and Buses.

As noted above, in discussing MCA it is usually taken that the weights used for summation reflect the preferences of the decision makers. However, it is clear that there is nothing inherent in MCA that would prevent those decision makers basing the weightings used on assessments of the preferences of the affected population, or at least taking account of those preferences in forming their own. The government could, for example, commission an opinion poll on the weights. We noted above that one of the objections that is raised against the use of the population's preferences in ECBA is that those preferences might well be based on inadequate information. Clearly, exactly the same objection could be raised in the present context. We also noted that Sagoff (1988, 1994) argues that the correct way to make decisions with serious environmental implications is through the *deliberations* of citizens. The point here is that Sagoff, and many others, think that individuals interacting and debating with one another will produce more informed preferences. We now note two ways that decision makers could use to get information on informed preferences.

11.3.5.3 Deliberative polling

Deliberative polling involves running an opinion poll then asking respondents to attend a meeting at which they will collectively consider the issues, by hearing and questioning expert witnesses and debating among themselves. At the end of this process, the participants are asked to again respond to the original survey instrument. As reported in Fishkin (1997) the results, in regard to the movement of opinion as between the first poll and that conducted after deliberation, are often striking. As an example, consider the results from three such exercises conducted in Texas. In Texas regulated public utilities are required to consult the public as part of their Integrated Resource Planning, and three chose to use deliberative polling to do this in regard to electricity supply planning. Respondents were asked to specify their first choice for the provision of additional power from four alternatives: renewable sources, fossil-fuel sources, energy conservation, buying in electricity. As between the two polls respondents attended meetings at which they were provided with, *inter alia*, cost data on these four alternatives. In each case there was the same pattern of response variation as between the before and after polls. As first choice, renewable sources fell from 67% to 16%, 71% to 35%, and 67% to 28%, while conservation rose from 11% to 46%, 7% to 31%, and 16% to 50%. The cost data showed conservation to be less expensive than renewable sources.

An obvious problem with deliberative polling is that it is very costly. The idea is to poll a random sample of sufficient size to produce results up to the standard usual in opinion polling. This may mean hundreds of people, which makes the information provision and deliberative parts of the exercise expensive, especially where the population of interest covers a large geographical area. As practiced to date, deliberative polling has usually involved opinions on somewhat broadbrush issues of interest to large media organisations. However, as exemplified by the example from Texas, the general strategy could, given funding, be applied to more narrowly defined decision problems, with respondents being required to consider resource constraints and their implications.

11.3.5.4 Citizens' juries

A citizens' jury exercise is less expensive than deliberative polling. In a report on experience with citizens' juries in the UK, Coote and Lenaghan (1997, p. ii) describe what is involved as follows:

> Citizens' juries involve the public in their capacity as ordinary citizens with no special axe to grind. They are usually commissioned by an organisation which has power to act on their recommendations. Between 12 and 16 jurors are recruited, using a combination of random and stratified sampling, to be broadly representative of their community. Their task is to address an important question about policy or planning. They are brought together for four days, with a team of two moderators. They are fully briefed about the background to the question, through written information and evidence from witnesses. Jurors scrutinise the information, cross-examine the witnesses and discuss different aspects of the question in small groups and plenary sessions. Their conclusions are compiled in a report that is returned to the jurors for their approval before being submitted to the commissioning authority. The jury's verdict need not be unanimous, nor is it binding. However, the commissioning authority is required to publicise the jury and its findings, to respond within a set time and either to follow its recommendations or to explain publicly why not.

Obviously the particulars described here are not immutable, and there could be considerable variation consistent with the underlying rationale.

In regard to underlying rationale, Coote and Lenaghan (p. ii, italics in original) put it as follows:

> Compared with other models, citizens' juries offer a unique combination of *information, time, scrutiny, deliberation and independence*.

Coote and Lenaghan report positively on the citizens' jury process. Of particular interest here, they judge that 'Jurors readily adopt a community perspective', that most 'accept that resources are finite and were willing to participate in decisions about priority setting', and that 'a substantial minority of jurors said they had changed their minds in the course of the session'. It should also be noted that a number of the participating jurors expressed 'strong doubts about the jury's capacity to influence the commissioning authority'. Experience in using citizens' juries in relation to decisions concerning the natural environment is limited; some references will be found under Further Reading.

11.3.5.5 Overview

In practice, decisions with serious environmental implications are generally taken by the management of private firms, or by politicians (or their delegated officers). The hope is that where private commercial decisions have serious environmental implications, those decisions will require in some way to be approved by government (or its delegated agency) taking the social welfare view of the options involved. We say 'the hope is' because in order for this to be the case law must be drafted so as to define the circumstances in which private decisions need governmental approval. It is in the nature of the case – imperfect knowledge of the future consequences of current action – that there can be no guarantee that all private decisions that will turn out to have serious environmental consequences will be subject to governmental, or other social, review. Equally, there will be circumstances in which public-sector projects which turn out to have serious environmental consequences do not get properly scrutinised.

Focusing on those cases where scrutiny is exercised, we have seen that the appraisal method preferred by economists has many critics, and that there are alternative methods. Basically all involve two stages. First, expert knowledge is used to estimate project impacts, then, preferences are used to reach a decision on the basis of the information about impacts. One argument often used in favour of ECBA is that the preferences that it uses are those of the affected general public rather than those of the experts. However, as we have seen, this 'democratisation' at the preference stage is not unique, or at least need not be unique, to ECBA. We have also, in this chapter and earlier chapters, seen that the efficiency criterion of ECBA does not necessarily imply that any particular concept of equity will be honoured in decision making, and does not guarantee sustainability. It is equally the case that MCA, for example, may, depending on the preferences used, produce decisions that are inefficient, inequitable, and inconsistent with sustainability. Advocates of deliberative processes have yet to demonstrate that these necessarily produce, for example, decisions consistent with sustainability requirements. It is also important to note that just as the practice of environmental valuation is subject to criticism and sometimes controversy, so is that of deliberative processes. Box 11.4 illustrates this for an exercise in deliberative polling.

Finally, we can note that those charged with reaching a decision on a project thought to involve serious environmental consequences do not have to regard ECBA, MCA and deliberative processes as mutually exclusive means for the provision of advice. In each case a necessary first step is the attempt to document all of the consequences of the project, or of each of the options under consideration. This is itself an important discipline on the decision-making process. There is then no reason why the impact assessment data should not be input to an ECBA, an MCA and, say, a citizens' jury. Unless these all produce the same result, a decision still has to be made by politicians or their appointed agents. All project appraisal methods should be regarded, as a practical matter, as providing information to decision makers rather than as providing them with the answer.

Box 11.4 Deliberative polling and nuclear power in the UK

As in most countries, the use of nuclear fission for electricity generation in the UK has been controversial, with strong advocates for and against. During the last two decades of the twentieth century, no new nuclear generating stations were built, and the electricity supply industry was privatised, so that the controversy was, to some degree, de-politicised. In the early years of the twenty-first century the UK government came to the view that, on account of the age of the existing nuclear plant, security of supply issues, and the climate change problem, it was necessary to revisit the question of whether new nuclear plant was desirable.

The UK government initiated a consultation process and subsequently, in July 2006, it issued a report giving its view that nuclear power had a continuing role in the electricity supply system, and that it would look favourably on projects to build new nuclear power stations. Consequent upon a legal challenge by Greenpeace, a longstanding opponent of nuclear power, the High Court ruled that the government's decision making process had been unlawful in as much as it had failed to engage in adequate consultation.

Following this decision, in May 2007 the UK government initiated a new consultation process, one element of which was a deliberative polling

Box 11.4 ***continued***

exercise. This took place in September 2007. The report on this exercise written by the market research firm, Opinion Leader, commissioned by the government to run it can be downloaded from the UK government website http://www.berr.gov.uk/files/file43548.pdf. It is interesting that on most questions, the change in the response percentages as between the initial and the final polls was small.

Greenpeace looked at the information provided to the participants and took the view that it was biased in favour of nuclear power. In October 2007, Greenpeace complained about the work of Opinion Leader to the Market Research Standards Board, MRSB, of the Market Research Society – Market Leader was a member of the Society which has a code of conduct for its members. The report of the MRSB is not now available on the Market Research Society website, but it was provided to Greenpeace in October 2008 and can be downloaded from http://www.greenpeace.org.uk/files/pdfs/nuclear/MRSfindings.pdf. It is instructive to read also the Greenpeace complaint itself, http://www.greenpeace.org.uk/climate/greenpeace-formal-complaint-to-msrc-over-nuclear-power-consultation, which details the ways in which it was alleged that participants were improperly informed.

The MRSB considered the Greenpeace complaint against B14 of the MRS code of conduct, which states that MRS members 'must take reasonable steps to ensure that Respondents are not led to a particular answer'. The MRSB found that Opinion Leader had not complied with B14, noting that 'deliberative research is a relatively new technique and that there are no current MRS guidelines on preparation or review of research materials specific to deliberative research'.

The UK government published *Meeting the Energy Challenge: A White Paper on Nuclear Power* (available at http://www.berr.gov.uk/files/file43006.pdf) in January in 2008, in which it drew on the results of the consultation exercise, and in which it stated its conclusion that 'it would be in the public interest to allow energy companies the option of investing in new nuclear power stations' and that 'the government should take active steps to facilitate this'. At the time that this box was written (January 2009) there had not been reported any further action by Greenpeace arising out of the MRSB finding on the deliberative polling exercise.

Summary

Intertemporal efficiency

Intertemporal efficiency requires the equality of consumption discount rates across individuals, of rates of return across firms, and that the discount rate is equal to the rate of return. An efficient allocation over time is not necessarily a fair one.

Intertemporal optimality

Intertemporal optimality obtains when an intertemporal social welfare function is maximised. The allocation is fair according to the criteria reflected in that function.

Markets and intertemporal allocation

Given the satisfaction of stringent conditions, the equilibria in markets that shift consumption over time will satisfy the conditions for intertemporal allocative efficiency.

Project appraisal using CBA

Given market failure, cost–benefit analysis is an application of welfare economics which is intended to select projects according to efficiency criteria. It uses the Net Present Value test, selecting projects for which NPV is positive.

Choice of discount rate

There is disagreement among economists both about how the discount rate to be used in CBA should be selected, and about what number should be used for a given approach to the selection.

Environmental CBA

Given that projects relating to the environment typically involve market failure, they generally require CBA.

Alternatives to Environmental CBA

Among those concerned for the environment there are many who consider that CBA is the wrong way to appraise projects that affect the environment, and alternatives have been proposed and used.

Further reading

The treatment of the derivation of the conditions for intertemporal efficiency in introductory and intermediate microeconomics texts usually leaves much to be desired. An exception is Varian (1987). Chapter 4 of Dasgupta and Heal (1979), which is an environmental and resource economics text, covers the basic issues. Dasgupta (1994) covers the basic optimal growth model with and without the use of a non-renewable resource as input to production. Many microeconomics texts do cover well the analytics of intertemporal behaviour by firms and individuals: see, for example, Gravelle and Rees (1981) and Hirshleifer (1980). Common (1995) uses numerical examples to work through the basic ideas in intertemporal efficiency, behaviour in the market for loanable funds, and optimal growth.

Most intermediate microeconomics texts discuss project appraisal according to both commercial and social criteria: see, for examples, Gravelle and Rees (1981), Layard and Walters (1978) and Varian (1987). Cost–benefit analysis as such is the subject matter of Pearce and Nash (1981) and Hanley and Spash (1993) – the latter is especially about environmental application. The introduction to Layard and Glaister (1994) is a good overview of all of the main issues in cost–benefit analysis – subsequent chapters in the book deal with particular issues in more detail, or review applications in particular areas.

Applications of cost–benefit analysis to environmental issues will be found in journals such as the *Journal of Environmental Economics and Management*, *Environmental and Resource Economics* and *Environment and Development Economics*. Government departments and agencies frequently carry out or commission cost–benefit analyses and publish the results, often making them available via the internet. The Innovative Strategies and Economics Group of the US Environmental Protection Agency makes many papers and reports available at the Economics and Cost Analysis website, www.epa.gov/ttn/ecas. See also the website of the National Center for Environmental Economics which can be reached via the links from the EPA site, or directly at www.yosemite.epa.gov/ee/epa/eed.nsf/pages/homepage?Opendocument. A useful way into UK government publications is to go to the website for the Department of Environment Food and Rural Affairs, www.defra.gov.uk, and search on cost–benefit analysis.

There is an extensive literature on the principles and practice of the choice of discount rate for cost–benefit analysis. Arrow *et al*. (1996) is the source of the distinction between the prescriptive and descriptive approaches that was introduced in Chapter 3. Stiglitz (1994) is an excellent account of the main issues; see also Harberger (1971), Heal (1981), Marglin (1963), and Broome (1992). Lind (1982) investigates discounting and risk in energy policy. Mikesell (1977) looks in detail at the practical choice of discount rates, and the implications of that choice. Price (1993) is a critique of the conventional wisdom on all aspects of discounting. Heal (2005) and Gintis (2000a) discuss the evidence that people use lower discount rates for the more distant future, and provide references to the original studies. Lyon (1990) goes into the shadow pricing of capital in some detail: the paper that first set out the international capital mobility argument against shadow pricing capital is Lind (1990).

The 1970s saw the publication of a number of books and papers concerning wilderness development. Many of these originated with the Washington DC organisation Resources for the Future, and involved the names Krutilla and Fisher. Particularly worth reading still are Krutilla and Fisher (1975) and Krutilla (1967). Porter (1982) provides an overview and synthesis of much of this literature. Resources for the Future publications span the whole range of resource and environmental economics, and can now be accessed at www.rff.org.

Foster (1997) is a collection of papers which cover most of the range of the ethical objections to environmental cost–benefit analysis, and discuss some alternative ways in which social decisions about the use of the environment might be made. The journal *Environmental Values* frequently includes papers about the ethical foundations of social choice in regard to the natural environment: Vol. 9, no 4 (November 2000) was, for example, a special issue on 'The accommodation of value in environmental decision-making'. *Ecological Economics* also frequently includes papers on these issues: see, for example, the special issue on 'Social processes of environmental valuation', Vol. 34, no 2, published in August 2000.

Glasson *et al*. (1994) is a very good introduction to the principles and practice of environmental impact assessment. Lee and Kirkpatrick (2000) is a collection of papers focusing mainly on the application of environmental impact assessment in developing countries. Janssen and Munda (1999) is a very useful brief survey of the principles and methods of multi-criteria analysis, on which see also Janssen (1992) for a fuller treatment. The UK Department for Transport, Local Government and the Regions produced (DTLR, 2001) a manual on multi-criteria analysis and its applications, which can be accessed at www.dtlr.gov.uk/about/multicriteria. Munda *et al*. (1994) is about qualitative assessment and Van Pelt (1993) relates multi-criteria analysis to sustainable development. Stirling (1997) makes a case for multi-criteria mapping as a means for informing decision making, rather than as a decision-making tool. The August 2000 issue of *Ecological Economics* includes papers on multi-criteria analysis and citizens' juries.

Jacobs (1997) reviews the arguments for citizen deliberation as the basis for environmental decision-making, discusses the role of citizens' juries and related procedures, and provides useful references. A useful source of information on participatory governance generally is http://www.peopleandparticipation.net/. The Centre for Deliberative Democracy at Stanford University has a useful website, http://cdd.stanford.edu, with accounts of exercises in deliberative polling. A similar centre at the Australian National University, http://deliberativedemocracy.anu.edu.au/index.php reports (February 2008) that it is starting a deliberative polling exercise on climate change. The Jefferson Center, in the USA, is a non-profit organisation interested in 'providing tools for decision makers to more fully understand what citizens want to do about key issues'. Information about its experiences with citizens' juries can be found at http://www.jefferson-center.org. In 2001, Vol. 19, no 4 of *Environment and Planning C* was a special issue on participation and deliberation in environmental valuation and decision making. Spash (2007a) discusses attempts to use participatory deliberation as a component of a process of monetary valuation, for CBA type use, and provides references to the emerging literature in this area.

Discussion questions

1. Should decisions about environmental policy be made on the basis of cost–benefit analysis?
2. In the context of a proposed hydroelectric development in a wilderness area, has the Krutilla–Fisher argument about the relative price movements that should be assumed in ECBA been affected by recent concerns about the implications for climate change of carbon dioxide emissions in fossil-fuel combustion and about nuclear power stations?

Problems

1. Derive the optimality conditions for the model specified in Appendix 11.1.

2. Consider the two social welfare functions

   $W_U = U(1) + U(2)$ (utilitarian)

   $W_R = \min\{U(1), U(2)\}$ (Rawlsian)

   where $U_i = \ln(X_i)$ is the utility enjoyed by the ith generation from the consumption X_i, $i = 1, 2$. Consider two projects:

   Project A: Generation 1 reduces consumption by 10 units. The investment yields 20 additional units of consumption for Generation 2.
   Project B: Generation 1 reduces consumption by 15 units. The investment yields 15 additional units of consumption for Generation 2.

   Let the pre-project level of consumption in Generation 1 be 100 units. Now consider three scenarios:

Scenario	Pre-project level of X_2
(i) No technology change	100
(ii) Technology improvement	120
(iii) Technology worsening (or loss of inputs)	80

   Use a tick to denote *Do project* or a cross to denote *Do not do project* in each cell of the following table to show whether the project (A or B) should be undertaken under each of the three scenarios, for the two cases of a utilitarian SWF (U) and a Rawlsian SWF (R).

		Scenario					
		(i)		(ii)		(iii)	
		U	R	U	R	U	R
Project	A						
	B						

3. The Safe Water Drinking Act required the United States Environmental Protection Agency to establish action standards for lead in drinking water. The EPA evaluated three options (labelled A, B and C below) using cost–benefit techniques. A selection of the results of this analysis is presented in the following table.

	Option		
	A	B	C
Total benefits	\$68 957	\$63 757	\$24 325
Total costs	\$6272	\$4156	\$3655
Benefit to cost ratio	11.0	15.3	6.7
Marginal benefit (MB)	\$5192	\$39 440	\$24 325
Marginal cost (MC)	\$2117	\$500	\$3665
MB to MC ratio	2.5	78.8	6.67

   Monetary values in the table are 1988 \$ million, based on a 20-year life, discounted to present value at 3%. Option A involves the strictest standard, Option C the least strict, with B intermediate. The marginal cost and benefit figures refer to incremental costs/benefits incurred in moving from no control to Option C, from Option C to Option B, and from Option B to A respectively.

   Source: EPA (1991). The EPA decision is discussed at length in Goodstein (1995), pp. 133–140.

   The US Environmental Protection Agency selected Option B. Is Option B the economically efficient choice?

4. Solve equation 11.31 for a with NPV set at 0 to get an expression for a^*, the value of a that makes the project marginal, in terms of r, X, P and D. Treat X, P and D as parameters and find $\partial a^*/\partial r$. What can be said about its sign? What is its sign for the values used in the chapter when discussing discount rate adjustment, $X = 1000$, $D = 75$ and $P = 12.5$? Confirm the answer by evaluating a^* for $r = 0.055$ and $r = 0.045$, and explain it.

5. Rework the MCA example considered in the chapter:

   a. With the following scoring of the qualitative assessment of the wildlife and amenity impacts

Bad	Moderate	Slight
1	2	3

   b. With the following scoring of the qualitative assessment of the wildlife and amenity impacts

Bad	Moderate	Slight
10	20	30

Is it possible to find an order-preserving scoring that affects the result of the weighted summation MCA?

CHAPTER 12 Valuing the environment

If the environment is one of the world's bloodiest political battlefields, economics provides many of the weapons. Environmental lawsuits and regulatory debates would be starved of ammunition if economists did not lob their damage estimates into the fray. The trouble with these number wars is that the estimate's accuracy is often more akin to that of second-world-war bombers than precision-guided missiles.

***The Economist*, 3 December 1994, p. 106**

Learning objectives

In this chapter you will

- learn about the categories of economic value assigned to the natural environment, and the distinction between use and non-use values
- work through the utility theory on which environmental valuation techniques are based
- learn how Contingent Valuation uses individuals' responses to hypothetical questions to infer use and non-use values
- find out about the technique of Choice Modelling
- find out how the Travel Cost method uses data on actual behaviour to infer use value
- learn about the Hedonic Price method
- be introduced to valuation methods that are based on production function analysis

Introduction

Deciding to exploit a natural resource typically entails an environmental cost. For example, deciding to quarry stone from a mountainside requires that society has to forgo not only the recreational values but also suffer adverse aesthetic impacts.

Unfortunately, it is frequently impossible to look to the market to provide us with direct information regarding the value society places on changes in the provision of environmental goods and services. The difficulty stems from the fact that so many environmental goods, such as polar bears, the ozone layer and attractive panoramas, are classic examples of public goods. Unlike with quarried stone or timber, both of which are bought and sold in the marketplace, there is no direct evidence on how much people are willing to pay for public goods such as the aesthetic appearance of the landscape because there is no possibility of withholding them. And even when it is possible to derive a demand curve for particular environmental goods, e.g. by charging admission to a recreational site that was previously free, it turns out that even this approach is liable to understate their value by not capturing all of the ways in which people benefit from them.

This chapter describes the techniques through which economists attach values to the unpriced services provided by the natural environment. We note that environmental valuation is a major activity for environmental economists.

The original, and still the principal, motivation for environmental valuation was to enable environmental impacts to be included in cost–benefit analysis.

As explained in the preceding chapter, the question of whether to allow a quarry to operate in an area used for recreational purposes would clearly require valuing the recreational benefits. Another purpose of environmental valuation is to determine the appropriate level for environmental taxes and pollution control standards (see Chapter 6). For example, what is the appropriate standard for concentrations of particulate matter in urban areas given epidemiological evidence that such pollution has adverse health impacts?

There are two further sources of demand for environmental valuations. One of these is the need to take account of environmental damage in measuring economic performance in national accounting exercises to be dealt with in Chapter 19. The other is the fact that, at least in the USA, since the late 1980s, economists' valuations of environmental damage are now admissible evidence in fixing the compensation to be paid by those whom the courts hold responsible for the damage. This has been a major stimulus to, and source of interest in, work on environmental valuation.

The basic strategy for environmental valuation is the 'commodification' of the services that the environment provides. These services are used by households and firms, and are treated as arguments in utility and production functions respectively. Standard theories of consumer and producer behaviour can then be used to devise methods for assigning monetary values to environmental goods and services. Most of the environmental valuation literature, however, is about services which flow to households rather than firms, and we shall follow that emphasis in this chapter.

The chapter is organised as follows. The first section presents a typology of the economic values that economists ascribe to environmental goods and services. The second section deals with the utility theory that underpins standard environmental valuation techniques. We then proceed to give extended accounts of a number of environmental valuation techniques. We begin by describing the contingent valuation method which involves asking people about their maximum willingness to pay for an environmental project, and then choice modelling which looks at how individuals trade off money against the attributes of environmental goods. The travel cost method infers recreational valuations from observed recreational trip behaviour, and the hedonic price technique looks at how the benefits of environmental quality are capitalised into land prices and wage rates. We conclude with a discussion of something called the production function approach, which seeks to value environmental inputs to production. More complex material on a valuation technique known as demand dependency is relegated to an Appendix.

12.1 Categories of environmental benefits

Individuals may derive value from environmental goods in many other ways than through direct consumption. If, therefore, we wish to value environmental resources properly we need to employ a much broader definition of value. This broader concept is known as total economic value (TEV).

TEV recognises two primary sources of value that individuals derive from the environment. These are 'use values' and 'non-use values'. Even within the class of use values it is useful to distinguish two further categories.[1] Taking a tropical forest as an example, consumptive use values would include those that come from exploitative uses such as the harvesting of timber, and collecting firewood and non-timber forest products e.g. latex and vegetable ivory used to make buttons. These environmental goods are destroyed in the act of using them. Many might be sold in the marketplace and therefore possess readily observable market prices. Valuing such goods is in principle straightforward.

Non-consumptive uses, by contrast, include using the forest for recreational purposes and for example the pleasure that individuals may derive from watching documentaries or reading articles on tropical forests presented to them through television, magazines or the internet. Such values arise from activities which are not necessarily destructive to the environment or necessarily rivalrous in

[1] The reader is warned that there is not in the literature a single standard categorisation, nor is terminology uniform.

nature.[2] However, in many cases it may be physically or legally impossible to prevent individuals from enjoying such benefits, in which cases non-market valuation techniques have to be employed. Frequently there is a tension between consumptive and non-consumptive uses of environmental assets.

The other major source of value is 'non-use values'. These refer to the benefits an individual may derive from a resource without ever physically interacting (directly or indirectly) with it or indeed ever even intending to use it. Individuals may quite simply derive satisfaction from the continued existence of particular species for no other reason than they find it repugnant that any species should be driven to extinction. Likewise, an individual may deplore the plan to build a new road through an archaeological site without intending to visit the site under threat. These are known as 'existence values' or 'intrinsic values'. In addition, so-called 'altruistic values' arise from a concern for human contemporaries. Even if an individual does not value a particular environmental good they may nevertheless value the satisfaction that other people obtain from using the resource. Finally, 'bequest values' arise through concern for the interests of future generations.[3]

We ought to mention that it is quite common to see 'option values' and 'quasi-option values' added to taxonomies of TEV. Such values arise only where there is incomplete knowledge of future conditions. These values are discussed in the next chapter which deals explicitly with risk and uncertainty. In this chapter we are assuming that the future is known.

Of course, for any particular project, and for particular individuals, some or all of the components of TEV may be zero. Typically the objective is simply to estimate TEV rather than its constituent components. Nevertheless, there is an important operational distinction between use values and non-use values. More specifically, whereas 'revealed preference' valuation approaches can estimate only use value, what are known as 'stated preference' approaches can be used to estimate both use and non-use values. Non-use values cannot be inferred from observed behaviour.

Before closing we note that many environmental goods, e.g. tropical forests and mangrove swamps, also perform a variety of functions that benefit individuals only indirectly. For example, tropical forests fix carbon from the atmosphere, regulate the microclimate and prevent soil erosion. One might therefore wish to distinguish between 'direct' and 'indirect' sources of value. Obviously it is important that both direct and indirect values of an environmental asset are taken into account in decisions about the use of the resource.

12.2 The theory of environmental valuation

In this section we will consider environmental damages affecting only consumers and defer any discussion of environmental damages suffered by firms until the very end of the chapter. What we seek is a monetary measure of the utility changes experienced on account of the environmental damage done by a project.

The first step in deriving appropriate monetary measures of the utility change associated with changes in the quality or quantity of environmental goods is the assumption that the quality or quantity of environmental goods can be treated as an argument in a well-behaved utility function.[4]

Let e represent the environmental good in question and y income (or spending on some composite marketed commodity). For example, e could be hectares of virgin forest or the concentration of dissolved oxygen in a lake where in both cases an increase in e is beneficial. We assume that the individual cannot adjust their consumption level of e.

Figure 12.1 represents the preferences of a given individual. The vertical axis measures the individual's income y, or equivalently the quantity of some

[2] Some recreation clearly damages the environment and some sites may be subject to congestion.

[3] There is some debate about whether values arising from altruistic motives ought to be included in cost benefit analyses, see McConnell (1997).

[4] This is an important first step as the conditions under which preferences can be represented by well-behaved utility functions are non-trivial, and, as we shall discuss later in the chapter, some commentators argue that preferences over 'environmental commodities' are unlikely, in many cases, to satisfy those conditions.

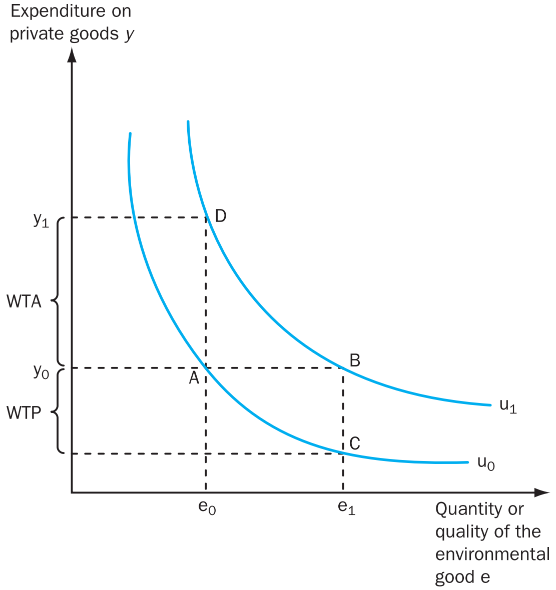


Figure 12.1 Measuring changes in welfare for an environmental good

regular composite commodity, whose price is normalised at unity, and the horizontal axis measures *e*. The utility *u* of the individual is given by the indirect utility function $u = u(y, e)$.

Two indifference curves u_0 and u_1 identify all combinations of the composite good and the environmental good between which the individual is indifferent. Indifference curves further from the origin represent higher levels of utility.

Consider the value to the individual of an increase in the quantity of the environmental good from e_0 to e_1. Suppose that initially the individual has y_0 private income and so is at point A on indifference curve u_0. Now look at point C which is also on the same indifference curve u_0. At point C the individual can enjoy e_1 of the environmental good but his income is reduced by the amount BC to stay on the same indifference curve. We can immediately infer that his maximum willingness to pay (WTP) for the increase in the level of the environmental good would be BC. This also corresponds to what economists call the compensating surplus (CS) for the increase in the level of the environmental good from e_0 to e_1.

Table 12.1 Monetary measures for environmental quality changes

	CS	ES
Improvement	WTP for the change occurring	WTA compensation for the change not occurring
Deterioration	WTA compensation for the change occurring	WTP for the change not to occur

Now consider the opposite case in which the individual, again starting with y_0 private income, suffers a decrease in the environmental good from e_1 down to e_0. The initial position of the individual is at point B. At point D, which is on the same indifference curve u_1, the individual enjoys only e_0 of the environmental good but with his income raised by an amount DA. We can infer that his minimum willingness to accept (WTA) compensation for the reduction in the environmental good is DA. This is the CS for the reduction in the environmental good e_1 to e_0.

Now let us consider briefly the two further measures. If the individual were at point A, what would be his minimum WTA compensation in lieu of an increase in the level of the environmental good from e_0 to e_1? This is once more given by distance DA. If the individual were at point B, what would be his maximum WTP to avoid a reduction in environmental quality from e_1 to e_0? This is given by the amount BC. These are known as equivalent surplus (ES) measures.

Table 12.1 summarises the situation in regard to monetary measures of utility change associated with changes in the quantity or quality of an environmental good *e*.

We will latterly go on to consider the extent to which these measures of value can differ from one another, along with the question of when one measure of value is more appropriate than another.[5]

[5] Whereas this section has discussed monetary measures of welfare change for households confronted by a change in the quantity of an environmental good, there are two corresponding monetary measures of the utility change associated with a price change. These are the compensating variation, which is the change in income that would 'compensate' for the price change, and the equivalent variation, which is the change in income that would be 'equivalent' to the proposed price change. The compensating variation is the quantity of money income which, when taken from the individual together with the price fall, leaves the individual at his or her initial level of utility. It is, therefore, the maximum amount that the individual would be willing to pay (WTP) to have the price fall occur. The equivalent variation is the quantity of money income which, if given to the individual without the price fall, would give the same level of utility as he or she would have attained if the price fall had occurred. It is, therefore, the minimum compensation that the individual would be willing to accept (WTA) in lieu of the price fall.

12.2.1 Empirical valuation techniques

The literature on empirical environmental valuation techniques and their application is now quite extensive. The first technique we shall look at is contingent valuation (CV). Then we consider choice experiments (CE). Together these two are known as 'stated preference' techniques. The chief characteristic of stated preference techniques is that they are based on survey data and that they enable researchers to measure both use and non-use values. And they are also able to estimate WTP or WTA measures of economic value. But they rely on obtaining reliable answers to hypothetical questions.

We then discuss a number of 'revealed preference' techniques including demand dependency (DD), the travel cost (TC) method and hedonic pricing (HP). To reiterate, the main limitation of revealed preference valuation techniques is that, being based on observed behaviour, they are able to measure only use values. But at least they exploit data on actual observed behaviour. Finally we consider techniques based on production functions appropriate when firms' production functions include the environment as an input.

12.3 Contingent valuation

CV is a survey-based valuation technique which involves asking a representative sample of the population questions about their WTP or WTA for environmental goods. CV can measure both use and non-use values whereas revealed preference methods capture only use values. This represents a key advantage for CV because for many environmental goods non-use values are arguably likely to be significant. CV also provides theoretically correct WTP and WTA measures of utility change.

CV is the most widespread of all environmental valuation techniques. CV studies have been conducted in many countries by agencies with responsibilities for environmental protection, government departments, international organisations, university researchers etc. Applications include valuing the benefits of improving air and water quality; assessing the benefits of outdoor recreation opportunities; measuring the preservation benefits of wilderness areas; valuing improvements in public utility reliability; measuring the benefits of reduced transport risks; and valuing the provision of sanitation and refuse collection services in developing countries.[6] Early examples of CV include Randall *et al*. (1974) and Brookshire *et al*. (1976).

The steps involved in conducting a CV study can be stated as follows:

1. Creating a survey instrument (i.e. questionnaire). This can itself be broken down into a number of tasks, including (a) identifying possible uses of and attitudes towards the environmental good in question, (b) constructing the hypothetical scenario, (c) deciding whether to ask about WTP or WTA, (d) determining an appropriate payment vehicle, (e) selecting an appropriate elicitation method, and (f) collecting auxiliary information about the respondent.
2. Choosing an appropriate survey technique.
3. Identifying the population of interest and developing a sampling strategy.
4. Analysing the responses to the survey.
5. Aggregating the WTP or WTA responses over the population of interest.
6. Evaluating *ex post* the success (or otherwise) of the CV exercise.

We will now discuss each of these steps in more detail before discussing some of the problems with CV and the enduring anomalies that have emerged.

12.3.1 Creating the survey instrument

CV surveys typically commence with a brief explanation of the purpose of the exercise and, if appropriate, assurances about the confidential or anonymous nature of the respondents' answers.[7]

[6] The Environmental Valuation Reference Inventory http://www.evri.ca/ assembled by Environment Canada provides an online database of thousands of CV studies.

[7] Sometimes it is desirable to secure the participation of individuals before informing them about the purpose of the study in order to prevent a problem of 'self-selection'.

The typical CV survey is likely to ask a series of questions relating to the use of the environmental good in question e.g. 'Have you heard of the site? How often do you visit the site? For what purpose(s) do you visit the site?' Attitudinal questions are also common. Respondents may be invited to identify the environmental problem(s) that most concern them. Respondents can also be asked whether they (a) strongly agree; (b) agree; (c) neither agree nor disagree; (d) agree or; (e) strongly agree with a series of statements, e.g. 'Environmental protection costs jobs'.

The purpose of such questions is primarily to generate a series of variables which can be used to check whether subsequent responses to key questions about WTP have a basis in individuals' attitudinal beliefs. For example, we would expect to find that individuals who answered '5' to the question 'How concerned are you about environmental issues on a 1 to 5 scale' should be WTP more to prevent tropical deforestation than individuals who answered '1'. We would also anticipate that someone who visited a threatened site on a frequent basis to pay more for its protection than someone who had never even heard of it prior to the survey.

The questionnaire would then go on to describe in detail the specific environmental problem. A CV study aiming to value the preservation of a particular species might supplement a verbal explanation by showing a photograph of the endangered animal and provide an estimate of the date by which the species will become extinct if nothing is done to prevent such an occurrence.[8] By contrast, a CV intending to value water quality in a river would probably provide a map helping to identify its location along with a description of the uses of the river, the species living in it etc.

Care must be taken to describe the situation in terms that the respondent can understand. It would, for instance, not be sensible to describe water quality by means of the concentration of dissolved oxygen or fecal coliform colonies per 100 mL. Instead, water quality should be described in terms of whether it is (a) suitable merely for boating; (b) suitable for fishing; or (c) suitable even for swimming.

Next some project is introduced which is intended to have environmental benefits either by improving environmental quality or preventing any further deterioration. The project must be carefully defined and credible to the respondent otherwise the entire CV exercise is jeopardised. In particular, some explanation is required to explain precisely how the change is engendered. In the context of the examples referred to above, this might involve initiating a breeding programme for an endangered species or building a municipal water treatment plant to tackle water pollution.

The survey should explain what difference the project would make to the respondent. For example, if the project involved cleaning a historical monument affected by deposits of soot then the respondent might be shown two sets of computer-generated photographs. The first set of photographs would illustrate the monument becoming progressively dirtier through time (the baseline). The second set of photographs would show the monument as it would look if a cleaning project were implemented (the counter-factual).

The payment vehicle describes the way in which the questionnaire will ask the respondent about his or her willingness to pay for the project. Payment is hypothetical. The payment vehicle typically involves increasing local or national taxes. Increases in utility bills (electricity or water service charges) are also common for projects aiming to improve air quality and water quality. Requesting voluntary donations to a fighting fund is, by contrast, not recommended. There is evidence that the choice of payment vehicle alters the resulting WTP (Rowe *et al*., 1980). The payment is usually defined as either a one-off payment or a recurrent charge per household per year. Whatever payment vehicle is proposed to the household it must be one to which all households in the population of interest are liable. For example, increasing the cost of fishing licences as a payment vehicle for a project intended to improve the water quality of a lake risks encountering some households who, despite holding environmental non-use values for the quality of the lake, never use it for fishing.

[8] The use of photographs requires careful pre-testing – see below.

WTP and WTA measures differ in that whereas the former would be expected to be limited by an individual's income, the latter is not so constrained. For either an improvement or deterioration, the individual can be asked about their WTP or WTA. For example, an individual can be asked about their WTP for an improvement in air quality or their WTA compensation in lieu of such an improvement. For a hypothesised reduction in air quality an individual can be asked about their WTP to prevent such a change or their WTA compensation for it taking place. But which is the correct question?

From our discussion in Section 12.2 it is clear that the answer to this question depends on the entitlements assumed. CS measures relate to the initial utility level and imply entitlements to the status quo. Thus, asking about WTP for an environmental improvement implies that the individual is entitled to the existing level, as does asking about their WTA compensation for any deterioration. ES measures relate to the new level of utility. Asking about WTA compensation in lieu of an environmental improvement implies an entitlement to the higher level, while asking about WTP to avoid an environmental deterioration implies an entitlement to the lower level.

It is often assumed that individuals have the right to current levels of environmental quality. Viewed from this perspective, improvements in environmental quality should be valued using WTP questions whereas reductions in environmental quality should be valued using WTA questions. Despite this, CV studies very often employ WTP questions regardless of the situation. The reason for this is that (a) respondents often struggle to answer questions about WTA compensation and (b) unlike WTP, WTA compensation is not bounded by income. Nevertheless, asking WTP questions about a good which the individual already believes they are entitled to can sometimes cause respondents to reject the scenario, e.g. 'Why should I have to pay to prevent a reduction in air quality in my neighbourhood? They should be paying me!'

The elicitation method refers to the manner in which WTP or WTA is extracted from the respondent. The most common elicitation methods are open-ended, bidding games, payment cards, referendum (also known as dichotomous choice), and double-bounded dichotomous choice. Although we will illustrate these methods in terms of WTP, they are all easily recast in terms of WTA.

Open-ended questions simply ask respondents 'What is your maximum WTP for this environmental project?' Open-ended questions have the advantage that they do not provide any implied cues about the worth of the environmental project and are, statistically speaking, very informative. On the other hand, open-ended questions are notoriously difficult for respondents to answer, leading to large non-response rates.

Bidding games involve asking respondents a sequence of questions such as 'Are you WTP £5 (for this environmental project)? Are you WTP £10? Are you WTP £20?' until the interviewer reaches a bid level at which the respondent says they are unwilling to pay any more. But bidding games risk anchoring, whereby respondents are influenced by the implied cue in the starting bid (Boyle *et al.*, 1986).

Payment cards (or ladders) present an individual with a card with different monetary amounts written on it. The individual is then invited to place a tick against those monetary amounts which they are certain that they would be willing to pay, and to place a cross next to those amounts that they are certain that they would not pay. This format has grown in popularity since it is easy for respondents and allows them to express a degree of uncertainty regarding the amount that they are WTP. There is always, however, a suspicion that the scale used on the card might influence the respondent's decision.

Single-bounded dichotomous choice referendum-type questions are viewed by many as the gold standard for CV. The respondent is asked to imagine that there is a referendum which, if passed, would implement the environmental project at a certain cost per household. The respondent is then asked whether they would support or oppose the project. The hypothesised cost per household (which in reality may bear no relation to the actual cost of implementing the project) is varied across the sample of respondents. One group of respondents might, for example, be told that the cost per household is \$10. Another group might be told that the cost is \$30 per household and a third told that the cost is \$100 per household. The bid design (the number and level of the bids) is typically informed by a small-scale 'pilot' survey. Without such information there is a risk that the study may not be very informative. It may turn out that the lowest bid level employed is too high for the majority of respondents or conversely that the highest bid level is too low.

When dichotomous choice questions are used the researcher does not observe directly respondents' maximum WTP. Instead the researcher observes whether the respondent's WTP is above or below the bid level. To estimate maximum WTP it is necessary to analyse the data using the econometric techniques described below.

Not only is this elicitation format simple for the respondent to comprehend, it is also thought to be incentive compatible (Hoehn and Randall, 1987). More specifically, it is in the interest of the respondent truthfully to reveal their preferences and not strategically to manipulate their response. Disadvantages include the fact that the dichotomous choice formats are not very informative. One knows only that the respondent is or is not WTP a particular amount and not their maximum WTP. This means that larger sample sizes are required, implying greater expense. Some respondents may engage in what is called 'yea-saying' when confronted with a referendum-type question. There is moreover evidence that WTP estimates can be altered by adding or deleting high bid values (Kriström, 1990).

A variant of the referendum question is the double-bounded dichotomous choice. In this payment format a respondent who agrees to pay the initial bid level is then confronted with a higher amount. Alternatively, if the respondent refuses the initial bid level they are asked about a smaller amount. The advantage of this technique is that statistically it is more efficient. More specifically, if a respondent answers 'Yes' to the first amount but 'No' to the second and higher amount then this bounds their WTP. Likewise a respondent may answer 'No' to the first amount but 'Yes' to the second and lower amount which also bounds their WTP. Statistical efficiency is an important consideration in view of the high cost of conducting surveys. See McLeod and Bergland (1999) for concerns regarding the double-bounded dichotomous choice model.

Different elicitation methods can result in different WTP estimates. In particular, it frequently appears that open-ended valuation exercises result in significantly lower values than referendum type questions. Of course, it is undesirable that the results of the CV experiments should depend on seemingly arbitrary choices about the elicitation format. However, the incentives for truthful preference revelation do differ between formats, and as a consequence one should expect the estimates to differ (with the binary discrete choice question predicted to yield truthful responses).

Immediately before answering the WTP question a good CV will remind respondents about the existence of substitute environmental goods: 'Even if the quality of site A is not improved remember that you can still visit site B which remains in pristine condition'. Likewise respondents will be reminded about their household income constraint and invited to think about what they would need to forgo in order to pay for the environmental project. The respondent is also told 'There are no right or wrong answers – we want to know what YOU think'.

Follow-up questions seek information on the motives underlying individual responses. In a referendum-type format individuals who agreed to pay the proposed amount will find themselves probed for an explanation. Those who refused to pay will also be asked for a reason and their responses used to identify any 'protest bids'.

Protest bids are WTP responses posted by individuals who seize the opportunity provided by the survey to protest against the agency responsible for implementing the project or rail against some irrelevant aspect of the survey, e.g. 'I am not WTP more tax for this environmental project because the government would in reality spend the money on something else'. Nevertheless, most CV surveys will encounter large numbers of individuals who are unwilling to pay anything either because they cannot afford to or because they are simply not interested in the environmental good. These are not protest votes.

Follow-up questions can also be used to identify individuals who have not paused to consider their household budget constraint, or who indicate that the sum offered is intended to pay for an environmental good other than the one intended by the researcher (often a more inclusive good). For example, an individual who agrees with the statement 'My bid was to save all endangered mammals' when the questionnaire referred only to a project to save the Sumatran Tiger should be removed from the survey.

The final section of the questionnaire collects information on respondent characteristics. These include age, gender, educational attainment, household income, and household composition as well as other things of potential relevance to explaining variation in respondent WTP such as their membership of environmental organisations and their geographical proximity to the site of the environmental project. Often the question about household income proves especially problematical and respondents need encouragement and a gentle reminder that the results of the survey are anonymous. The questionnaire should also make clear precisely what is meant by 'household income'.

After each interview the interviewers themselves are debriefed. They are asked how much confidence they have in the respondent's answers, whether the respondent appeared to understand everything that they were asked to do and whether the interviewee appeared to take the interview seriously.

Enormous effort goes into designing CV surveys. The questionnaire is typically first tested in a focus group setting. Focus groups are small groups of individuals who are led through a loosely structured discussion of the issues raised by the environmental project and associated payment vehicle. Following any revisions the survey is then tested in the field on anything between 25 and 100 individuals. The final survey itself might be between 200 and 1000 individuals or even more depending on the desired accuracy.

12.3.2 Choice of survey method

There are three broad options for distributing the survey questionnaire.

Conducting face-to-face interviews is extremely expensive but offers several potential advantages, notably a high response rate. Face-to-face surveys can be conducted either in the respondent's home or in some other location such as the site being valued.

Mail surveys are much more economical, but get far lower response rates and tend to limit the amount of information that can be provided and the number of questions that can be asked. There is no control over the identity of the respondent or over the sequencing of questions. There is also the problem of self-selection bias (only those with strong feelings tend to reply to unsolicited mail-shots).

Telephone interviewing is cheap, but restricts the information that can be provided. Nevertheless, these techniques are not mutually exclusive and questionnaires can be mailed out to respondents who are subsequently contacted by telephone.

Which of these methods is to be used will clearly influence the design of the survey questionnaire. And in some circumstances one or more of these methods might be infeasible. For example, in CV experiments where the provision of photographic material is necessary, telephone surveys cannot be used, at least not in isolation.

12.3.3 Sampling

The first step is to identify the population from which the sample is to be drawn. Researchers need to identify that group liable to be affected by a change in the level of provision of the environmental good.

If we are concerned only with use values and if the environmental good has an obvious user group then the population from which one should draw a sample is already well defined. For example, if we wished to assess the use value of improved water quality in a lake, an obvious user group would be the population of anglers using the lake. Of course, individuals who do not directly use a resource may nonetheless value it. Sampling from a much larger population, e.g. all households within a certain radius of the site, would permit researchers to assess non-use as well as use values. In some circumstances the relevant population might be defined by the political boundaries of the Local Authority whose residents will be required to pay for the project intended to deliver environmental benefits if it is implemented. For certain non-use values, e.g. those relating to tropical forests, the candidate population is potentially the whole global population.

The required sample size will be determined by a number of considerations, including the statistical precision required and the financial cost of the surveys. Some elicitation methods can generate a high number of refusals and some, like the double-bounded dichotomous choice, garner more information than others. Frequently the sample of respondents is split in order to examine whether a differing treatment, e.g. in terms of the information provided or the scope of the project, affects the WTP estimates that emerge, increasing further the minimum sample size.

Above all, it is vital that the sample is representative of the target population. All members of the relevant population should have an equal probability of being included in the sample. This might be achieved by comparing the characteristics of the sample with the characteristics of the population as revealed by a national census.

12.3.4 Analysing the responses

Having conducted the CV survey the next step is to analyse the data in order to determine statistics of interest. These are typically mean and median WTP. The treatment of outliers and protest responses can have significant implications, especially for mean WTP. In what follows it is customary to calculate mean and median WTP excluding protest bids or other 'problematic' responses.[9]

The appropriate method of analysing the data depends on whether they were generated in response to open-ended 'What is your maximum WTP?' type questions or closed-ended referendum-type questions. Analysing open-ended WTP data to obtain the mean or median WTP is sufficiently straightforward to require no comment and is a task best undertaken using a spreadsheet package.

The analysis of closed-ended data is more involved. There are two methods of analysing dichotomous choice referendum-type data, known as the Random Utility Model (RUM) and the random expenditure function. Both are examples of what are known as 'parametric' estimators since they involve making certain distributional assumptions, e.g. that WTP is normally distributed. A full treatment can be found in Hanemann and Kanninen (1999). Afterwards we will present non-parametric estimators that avoid making any distributional assumptions.[10]

The RUM approach to analysing dichotomous choice referendum data assumes that the respondent votes 'Yes' if the utility provided by the package, including the provision of the environmental good combined with increased tax liability, exceeds the utility from saying 'No'. In other words, the respondent votes 'Yes' if

$$v(\beta(y - t) + \alpha x) + \varepsilon > v(\beta y) + \varepsilon \tag{12.1}$$

Where v is the individual's (linear) indirect utility function, y is the respondent's income, t is the tax level, x is a dummy variable which takes the value unity if the environmental good is provided and is zero otherwise, and ε is an independent and identically distributed random error term. If the random error term ε follows a Gumbel distribution then it can be shown that the probability of an individual saying 'Yes' is

$$\Pr(Yes) = \frac{e^{(\alpha-\beta t)}}{1 + e^{(\alpha-\beta t)}} \tag{12.2}$$

This is the Logit model with a constant term α and a single regressor, t, with coefficient β. If we had assumed instead that the error term followed the normal distribution then the standard Probit model would emerge. The parameter values are calculated using maximum likelihood techniques. Any standard statistical software package will estimate the Logit and the Probit model so obtaining parameters α and β is easy.[11]

The parameter α represents the extra utility generated by the provision of the environmental good whereas the parameter β represents the (negative of the) marginal utility of income. The coefficient on the bid variable t ought to be negative, indicating that as the implied tax liability increases the probability of saying 'Yes' will decline, whereas the constant term parameter α should be positive. Irrespective of whether the data have been analysed using the Logit or the Probit model, mean (and median) WTP is given by

[9] We note in passing that a CV study will typically provide a lot of other information besides WTP values, some of which might be of considerable interest to those in charge of managing a site of recreational interest.

[10] This section draws very heavily on Haab and McConnell (2002).

[11] A good primer in the Logit and Probit model is Gujarati (2003).

$$E[WTP] = -\frac{\alpha}{\beta} \tag{12.3}$$

Empirically it seems to matter very little whether the Logit or Probit model is used. It is further possible to include socio-economic variables by interacting them with the dummy variable indicating whether the environmental good is provided or not.

The second theoretically consistent approach to modelling WTP is called the 'random expenditure function' approach. The respondent votes 'Yes' to the WTP question when their 'internal WTP' exceeds the tax level and 'No' otherwise. In other words, the respondent votes 'Yes' when

$$WTP(y, x, \varepsilon) > t \tag{12.4}$$

where y, x and ε are as before. Selecting a particular functional form for the WTP function we might choose

$$WTP = e^{\alpha-\varepsilon} \tag{12.5}$$

Where α is a parameter whose value we wish to estimate. One consequence of using this functional form is to constrain WTP to be non-negative. If we were not concerned about this restriction then we could equally as well choose a linear functional form in which case the estimation of WTP would be identical to that in the RUM approach. We emphasise, however, that the difference between the RUM approach and the expenditure function approach is nothing to do with whether one assumes a linear or an exponential WTP function. The calculations in the random expenditure function approach can also be undertaken using either the Logit or the Probit model.

The probability that this function exceeds the tax level is given by

$$\Pr[WTP > t] = \Pr[e^{\alpha-\varepsilon} > t] = \Pr[\alpha - \varepsilon > \ln(t)] = \Pr[\alpha - \ln(t) > \varepsilon] \tag{12.6}$$

If we now make the assumption that the random error term ε is normally distributed with zero mean we can transform the model by dividing through by the standard error

$$\Pr[\alpha/\sigma - \ln(t)/\sigma > \theta] = F[\alpha/\sigma - \ln(t)/\sigma > \theta] \tag{12.7}$$

where F is the standard normal cumulative distribution function. Note that the coefficient on $\ln(t)$ is $1/\sigma$. The objective is to estimate the parameters α and σ. The first of these can be obtained by dividing the coefficient on the constant term in the Probit regression by the coefficient on $\ln(t)$. At this point the similarity with the RUM approach should become apparent. Calculating mean WTP is slightly more involved in the case where $\ln(t)$ rather than t is used as a regressor

$$E[WTP] = e^{\alpha} E[e^{\varepsilon}] = e^{\alpha+\sigma^2/2} \tag{12.8}$$

Any intermediate-level statistical text will verify that this is the mean of a lognormal distribution whereas

$$WTP_{median} = e^{\alpha} \tag{12.9}$$

Accordingly, when $\ln(t)$ rather than t is used in the Probit regression the mean and the median are no longer identical.

Although ubiquitous, the use of the Logit or Probit model to estimate WTP from dichotomous choice data has been criticised because of the inherent distributional assumptions. There is no *a priori* reason to assume that WTP possesses a particular distribution.

An alternative is to use non-parametric estimators which impose no restriction at all on the distribution of WTP. The Turnbull estimator is one such technique which can be applied to referendum-type data. The Turnbull estimator makes no assumptions other than that as the tax or bid level increases the proportion of people who are willing to pay should monotonically decline. For an example of the Turnbull estimator applied to referendum-type contingent valuation data, see Haab and McConnell (1997). We now outline the steps involved in analysing referendum-type data using the Turnbull estimator. The same estimator can also be applied to double-bounded dichotomous choice data if the response to the second bid level is ignored.

1. For bids indexed $j = 1, \ldots, M$ calculate $F_j = N_j/R_j$ where N_j is the number of 'No' responses to bid level t_j and R_j is the total number of responses.
2. Beginning with $j = 1$, compare F_j and F_{j+1}. The proportion of 'No' responses should increase as the bid level increases.
3. If indeed $F_{j+1} > F_j$ then continue.

4. If $F_{j+1} \leq F_j$ then pool cells j and $j+1$ into one cell with boundaries (t_j, t_{j+2}) and calculate $F_j^* = (N_j + N_{j+1})/(R_j + R_{j+1}) = N_j^*/R_j^*$.
5. Continue to pool adjacent cells until the Cumulative Density Function is monotonically increasing.
6. Set $F_{M+1}^* = 1$ and $F_0^* = 0$.
7. Calculate the Probability Density Function as the step difference in the final Cumulative Density Function: $f_j^* = F_{j+1}^* - F_j^*$ for each bid level.
8. Multiply each bid level (t_j) by the probability f_j^*.
9. Sum the quantities from the preceding step over all bid levels to get an estimate of the lower bound on mean WTP

$$\text{WTP } E_{LB}(WTP) = \sum_{j=0}^{M} t_j f_j^*$$

10. The highest bid level at which the $F_j^* \leq 0.5$ provides a lower bound on median WTP.

Despite the simplicity of the Turnbull estimator it possesses a number of disadvantages. These include the need to have larger sample sizes and the fact that only lower-bound estimates are available. It is impossible to use the Turnbull estimator to observe how WTP varies with household characteristics (short of stratifying the data) or indeed to adjust WTP estimates if the sample of respondents is atypical of the population.

Box 12.1 An example using the Turnbull estimator

The following referendum data (Table 12.2) describe WTP to create a nature reserve. The first two columns list the bid levels ranging from £0.00 up to infinity and the number of respondents interviewed. The third column lists the number of individuals refusing to pay the respective bid amount. Looking down the column, the number of refusals grows as the bid level increases but not monotonically so, and this is the problem that the Turnbull estimator seeks to address.

The fourth column calculates the probability of refusal. The probability of refusing a bid level of £0.00 is obviously 0 and the probability of refusing a bid level of infinity is obviously 1. In the fifth column the cumulative density function (CDF) is smoothed so that it is monotonically increasing. This is generally the same as column immediately to the left except where the probability is not monotonically increasing, such as for the bid level of £10.00. In this case the bids for £4.00 and £10.00 are pooled together conservatively, implying that 25 out of 100 respondents would refuse a bid level of £4.00. Hence the value entered alongside the bid level of £4.00 is 0.25. The remaining entries in the smoothed CDF column are identical to those in the Probability column. Column six presents the smoothed probability density function obtained by differencing the vertically adjacent cells in the smoothed CDF column.

Having completed the table it is now possible to calculate non-parametric mean and median WTP. The lower bound to median WTP is £20.00. The next highest bid level of £50 is rejected by 70% of individuals. The true median lies between £20.00 and £50.00. The lower bound on mean WTP is £26.90 and this is calculated by multiplying the PDF with the bid levels and then summing the final column. In other words we have £0.00 × 0.1 + £2.00 × 0.15 + £4.00 × 0.15 + £20.00 × 0.3 + £50.00 × 0.2 + £100.00 × 0.1 = £26.90.

Table 12.2 An example of the Turnbull estimator

Bid (t)	Respondents (R)	Refusals (N)	Prob. (F)	Smoothed CDF (F^*)	PDF (f^*)	$t \times f^*$
£0			0.0	0.0	0.1	£0.00
£2	50	5	0.1	0.1	0.15	£0.30
£4	50	15	0.3	0.25	0.15	£0.60
£10	50	10	0.2	Pooled	–	–
£20	50	20	0.4	0.4	0.3	£6.00
£50	50	35	0.7	0.7	0.2	£10.00
£100	50	45	0.9	0.9	0.1	£10.00
Infinity			1.0	1.0	–	£0.00
Total						£26.90

12.3.5 Mean or median willingness to pay?

We have seen how it is possible to estimate both mean and median WTP. Most research papers provide estimates of both. Where the mean and the median are allowed to differ it is often observed that the mean value is higher, sometimes by a considerable margin. Which is more appropriate for policy purposes?

Although economic theory suggests that mean WTP is more appropriate, there are two practical arguments in favour of using the median. First, there is the issue of political acceptance. We are not here suggesting that researchers should in any sense tailor their results to make them more acceptable, but merely pointing out that in order to gain democratic support it may be necessary to base any decision on a value for the environmental good which at least 50% of households would be WTP. The second reason is that the median may be more robust to outlying observations. We have already noted that in many CV exercises there are a handful of individuals who submit very high bids, failing, perhaps, to take into account their household's budget constraint. These should, but might not always, be flagged up as problematic bids. Although few in number, such individuals can exert considerable influence on estimates of mean WTP.

12.3.6 Aggregation

The final task of any CV exercise is to aggregate mean or median WTP values over the target population. Sometimes the target population will be the national population but, as discussed, it might be a smaller population such as the number of people using a particular environmental resource or the number of households living within a particular radius of the site or living in a particular political region.

If the sample is representative of the target population then aggregate WTP is simply mean or median WTP multiplied by N, the number of observational units (typically the number of households) in the target population

$$Aggregate\ WTP = N \times WTP \tag{12.10}$$

If the sample is not representative of the target population then the appropriate procedure is either to stratify the sample and calculate a mean WTP appropriate for each household type or to predict an appropriate value using statistical regression in order to achieve the same goal. Mean WTP for each household type is then multiplied by N_i, the number of observational units of each type present in the target population

$$Aggregate\ WTP = \sum_i N_i \times WTP_i \tag{12.11}$$

This procedure is especially important if the sample is geographically unrepresentative and WTP for the project depends on distance to the site.

12.3.7 The reliability and validity of contingent valuation surveys

Establishing the success or failure of a CV study involves weighing evidence of various kinds but there is usually no entirely satisfactory way of validating the results.

CV surveys ought to be reliable in the sense that administering the same survey to a different sample of respondents or the same sample of respondents at a later date should yield similar results. See Loomis (1989) for an assessment of test–retest reliability. Surveys designed and undertaken by different researchers but purporting to measure the same thing should also produce similar results.

When researchers talk about 'face validity' they mean whether a survey has asked the right question and specified the right scenario. Results from surveys characterised by a large number of protest bids or no-response items should be viewed with scepticism. Estimates from studies with vaguely described goods and vaguely defined payment obligations should be viewed likewise. How sensitive is average WTP if recalculated following a different protocol in relation to outliers and protest bids? Is the scenario credible to the respondent? Is the sample representative of the target population? Clearly these are not objective tests and even experts might disagree about the face validity of a particular CV study.

Criterion validity involves comparing the results of a CV study with actual market prices. Of course, the reason for doing a CV study in the first place is the fact that market prices are not generally available for non-excludable public goods such as environmental quality. Some researchers have used a CV survey with a dichotomous choice valuation question to predict the outcome of an actual referendum to determine spending on measures to improve environmental quality (Johnston, 2006).

Convergent validity asks whether the results of the CV study coincide with those obtained by another valuation method. Of course, such convergence does not definitively establish that the CV result is correct. Carson *et al*. (1996) find that in their study of over 600 comparisons CV estimates are on average lower than revealed preference estimates.

Theoretical validity can be established by estimating a 'bid function' (or regression equation) in order to ascertain whether various influences on respondents' WTP correspond to plausible (sometimes theoretical) expectations. In any bid function one would hope to find WTP negatively related to the bid level as well as distance from the site of the environmental project. Put differently, as the hypothesised price to the household of providing the environmental good in question rises the number of households supporting the environmental project should decline. And the greater the distance from the site the less likely it is that any member of the household will use the site and hence care about it.[12] WTP should, on the other hand, be positively related to income and the quantity of the environmental good being provided.

If the estimated parameters are consistent with theory and expectation some confidence may be placed in the survey results. But a bid equation lacking satisfactory explanatory power and coefficients without the expected signs suggests that the WTP responses are random and completely useless.

12.3.8 Types of bias and problems in contingent valuation

Numerous potential biases and anomalies have been identified in the CV literature. Good questionnaire design and testing is the way to detect and minimise these problems.

The first example of a bias is where the environmental good valued by the respondent differs from that intended by the CVM analyst. 'Part-whole' bias occurs when the value for a specific good is identical to the value for a more inclusive good. For example, a WTP question might refer to protecting a particular species from extinction. But the respondent's response might be intended to rescue all endangered species. Evidence to support such 'embedding' is presented by Kahneman and Knetsch (1992).

One problem with CV is the hypothetical nature of the exercise. Some researchers have used experiments to compare actual and stated values for a range of goods. Surveying these experiments, Harrison and Rustrom (2005) find evidence of hypothetical bias ranging from 2% to 2600%. One response to the problem of hypothetical bias is explicitly to ask individuals to avoid responding in such a manner! This is called 'cheap talk' (see Cummings and Taylor, 1999).

Insensitivity to scope occurs when an individual's WTP appears independent of the scale of the project presented to them. This might be interpreted as a manifestation of part–whole bias or alternatively that the bids made were largely symbolic in nature.[13] Internal tests of scope compare WTP for projects of differing scale presented to the same individual. External scope tests are where different groups of individuals are asked to value projects of a different scale and their responses compared afterwards. See Heberlein *et al*. (2005) for a further discussion.

[12] It is less clear whether existence values should be expected to decline with distance from the site.

[13] In one of the original papers calling attention to this problem it took the form that WTP for cleaning up the lakes in one region of the Canadian province of Ontario was 'strikingly similar' to that for cleaning up all the lakes in that province. In another CVM application it was found that WTP to prevent bird deaths did not differ significantly across scenarios in which the programme prevented 2000, 20 000 and 200 000 deaths. Carson (1997) reports results from over 30 CVM applications where the hypothesis of scope insensitivity can be tested and is rejected. He argues that where there is scope insensitivity it is a consequence of 'poor' survey design and administration. Others take the view that, at least in some cases, these insensitivities reflect problems with the behavioural assumptions underlying the CVM.

Warm-glow effects arise in a situation where a respondent derives satisfaction from making a symbolic commitment to a cause. Such behaviour may stem from a desire to gain status in the eyes of the interviewer.

Interviewer bias occurs when the identity of the interviewer impacts on the WTP responses typically obtained.

Prominence bias occurs if the very fact that an interview is taking place leads the respondent to conclude that the environmental good must be of somewhat greater significance than he or she had hitherto imagined.

Temporal embedding refers to a situation in which WTP does not vary with respect to the frequency of payment. For example, one would expect that a respondent's maximum WTP as a one-off payment would be much higher than their maximum WTP as an annual or monthly payment. In some applications, however, these amounts appear to be disturbingly similar. At best, this suggests that respondents have a surprisingly high discount rate.

Strategic bias refers to a situation in which respondents believe that their stated WTP will subsequently be used to determine their individual tax liability. Hence there is a motive to understate one's WTP (thereby free-riding on the contribution of others). One reason why CV has evolved from using the open-ended bid format to the dichotomous choice model is the theoretical result that the latter is immune to strategic bias.

Starting-point bias occurs when respondents anchor on values presented to them in the course of a survey. This is most likely to occur in a bidding game in which the starting point of the bidding game influences the maximum amount that individuals seem WTP. But it could also affect dichotomous choice valuation studies. This is an argument in favour of using open-ended WTP questions.

Yea-saying refers to a situation in which individuals responding to close-ended questions administered in a face-to-face or telephone interview say 'Yes' simply in order to please the interviewer. This leads to exaggerated estimates of WTP. There may also be the problem of nay-saying in some instances, particularly with the double-bounded dichotomous choice format where someone who agrees to a particular bid level is suddenly, and unexpectedly, confronted by another, much higher bid level.

Sequence effects refer to a situation in which the individual is asked about their WTP for a series of environmental goods, and it is discovered that WTP for individual goods is dependent upon the precise sequence in which these goods are presented to the respondent (Bishop and Welsh, 1992). This is sometimes referred to as the 'adding-up' problem. In fact this may not be a bias at all, especially if these environmental goods are substitutes for one another. The budget constraint of the individual also changes as they 'purchase' additional environmental goods.

Information bias refers to a situation in which an individual's WTP reflects the inadequacy of their knowledge. Once again this is, arguably, not a bias as such but nevertheless a reminder that there are some instances in which the views of informed experts should weigh more heavily than those of the general public. Information 'bias' occurs when an ecosystem performs important indirect functions but these functions are not properly appreciated by, or explained properly to, a relatively uninformed CV respondent who consequently 'undervalues' the ecosystem. For a systematic review of information bias and an example, see Hanley and Munro (1994).

12.3.9 The WTP/WTA disparity

One of the most enduring puzzles of CV is the disparity between WTP and WTA values. Theory predicts that WTA should exceed WTP by a small amount. To see why, look again at Figure 12.1 and consider the case in which the individual starts with private consumption of y_1. In this case, WTP for an increase in the environmental good from e_0 to e_1 is DA which is greater than BC. If the individual is richer, he is typically willing and able to spend more in order to increase the level of the environmental good. But notice also that the difference between y_1 and y_0 is the same as WTA for a reduction from e_1 to e_0. Large divergences between WTA and WTP are therefore possible only if an individual's WTP would change significantly if his income were augmented by an amount equal to WTA. Such an effect is likely to occur only if (a) WTA accounts for a significant share of the individual's income and (b) WTP is highly income elastic.

As an example of the orders of magnitude involved, suppose that each 1% increase in income leads to a 1% increase in WTP. Then if WTA is equal to 1% of an individual's income, the ratio of WTA to WTP is 1.01. But in most applications WTA will be much less than 1% of income. And most empirical evidence appears to suggest that the income elasticity of WTP for environmental goods is quite low as well. Together this suggests that the difference between WTP and WTA should be small. But in rather many studies WTA frequently exceeds WTP by an order of magnitude or more.[14]

Hanemann (1991) showed that utility theory actually predicts that for commodities where there are limited possibilities for substitution, WTA could be much larger than WTP. In the limiting case where there is perfect substitution between the composite commodity and the environmental good, the indifference curves become straight lines and the difference between WTP and WTA disappears. Other explanations of the WTP/WTA disparity include the possibility that this is a consequence of individual's lack of familiarity with WTP/WTA questions.

Another important explanation of the divergence between WTP and WTA is provided by 'prospect theory', outlined in a much-cited paper by Kahneman and Tversky (1979). Prospect theory has three essential aspects. First, an individual views things in terms of changes from the reference level, which usually corresponds to that individual's status quo. Second, the values of the outcomes for both gains and losses exhibit diminishing returns. Third, there is 'loss aversion'. This means that the value function is steeper for losses than for gains. This gives rise to an 'endowment effect' whereby people value a good or service more once their property right to it has been established. People place a higher value on things that they already own than things that they do not. If the value function v is a function of the change in the level of the environmental good x then $v(x) = x^{\alpha}$ if $x > 0$ and $v(x) = -\lambda(-x^{\alpha})$ if $x < 0$ (with a typical $\alpha = 0.88$ and $\lambda = 2.25$). In the view of many, prospect theory represents a great improvement over expected utility theory. Many observed violations of expected utility theory are predicted by prospect theory. The movable reference point in particular is the major difference between prospect theory and traditional economic theory.

If WTP and WTA answers can be very different then it matters which one is used, since it could make the difference between approving and rejecting a project. But as already discussed, the choice between WTP and WTA is really a decision about property rights. If property rights are considered to be defined by the current level of environmental quality then improvements in environmental quality should be valued using WTP and reductions in environmental quality should be valued using WTA.[15]

Box 12.2 The NOAA panel

In the USA the courts have decided that CV-based evidence may be admissible in determining the compensation payments to be made where actual damage has occurred. The sums involved can be large and this has sharpened the controversy. The US government agency responsible for setting the rules for the assessment of damages from oil spills, the National Oceanic and Atmospheric Administration of the US Department of Commerce, convened a panel of experts, co-chaired by two Nobel laureates in economics (Kenneth Arrow and Robert Solow), to advise on the reliability of CV for the role allowed it by the courts.

The Panel's report (US Department of Commerce, 1993) gave CV for what it referred to as passive-use (i.e. non-use) values a qualified endorsement. It states that: 'The Panel starts from the premise that passive-use loss – interim or permanent – is a meaningful component of the total damage resulting from environmental accidents'. It further comments that: 'It has been argued in the literature and in comments addressed to the Panel that the results of CV studies are variable, sensitive to the details of the

[14] In the first study to estimate both, for example, Hammack and Brown (1974) reported WTA four times larger than WTP for the same change.

[15] Experience with CV indicates that many individuals have difficulty with WTA questions.

Box 12.2 *continued*

survey instrument used, and vulnerable to upward bias. These arguments are plausible. However, some antagonists of the CV approach go so far as to suggest that there can be no useful information content to CV results. The Panel is not persuaded by these extreme arguments.' The Panel went on to say that: 'The simplest way to approach the problem is to consider the CV survey as essentially a self-contained (sample) referendum in which respondents vote on whether to tax themselves or not for a particular purpose.'

The Panel identified a number of stringent guidelines for the conduct of CV studies, concluding that: '. . . under those conditions . . . CV studies convey useful information. We think it is fair to describe such information as reliable by the standards that seem to be implicit in similar contexts, like market analysis for new and innovative products and the assessment of other damages normally allowed in court proceedings.' The Panel also stated that: 'CV studies can produce estimates reliable enough to be the starting of a judicial process of damage assessment, including lost passive-use values.'

The Panel's guidelines covered all aspects of the design and conduct of a CV study. The *Exxon Valdez* exercise can be considered as exemplifying compliance with the guidelines. In particular, the Panel recommended:

1. Probability sampling from the entire affected population
2. Minimise non-responses
3. Personal interviews
4. Careful pretesting for interviewer effects
5. Clear reporting, of defined population, sampling method, non-response rate and composition, wording of the questionnaire and communications
6. Careful pretesting of the CV questionnaire
7. Conservative design (prefer options that tend to underestimate, rather than overestimate, WTP)
8. WTP format instead of WTA
9. Referendum format
10. Accurate description of program of policy
11. Pre-testing of any photographs to be used
12 Reminder of undamaged substitute commodities
13. Adequate time lapse from any incident to be valued
14. Temporal averaging of responses
15. 'No-answer' option available
16. Yes–no follow-ups to referendum question
17. Cross-tabulations with other questions such as attitudes toward site, environment etc.
18. Checks for understanding
19. Alternative expenditure possibilities provided
20. Present-value calculations made as clear as possible

Box 12.3 Using CV to estimate damages from the *Exxon Valdez* oil spill

In 1989 the *Exxon Valdez* ran into submerged rocks shortly after leaving the port of Valdez loaded with crude oil, and 11 million gallons of its cargo flowed from ruptured tanks into the waters of Prince William Sound off the coast of Alaska. This was the largest oil spill in US waters, and was widely regarded as a major environmental disaster, occurring as it did in a wilderness area of outstanding natural beauty. In anticipation of legal action against the ship's owners, the Government of Alaska commissioned a team of economists to conduct a CV study to estimate the damages from the oil spill. In terms of the steps in doing CV work identified and discussed in the text, what the research team did can be summarised as follows.

In designing the scenario and selecting a payment vehicle, the team's primary goal was to develop a survey instrument that would produce a valid and theoretically correct measure of the lost values due to the natural resource injuries caused by the spill. This was seen as entailing:

(a) a scenario which fully described the spill's impacts and was intelligible to all potential respondents;
(b) a plausible payment vehicle;
(c) a scenario that would be seen by respondents as neutral as between the interests of the government, the oil company and environmentalists;
(d) a conservative approach to scenario construction, erring on the side of understating the environmental effects of the oil spill.

Box 12.3 *continued*

Requirement (a) was taken to imply the extensive use of maps, colour photographs and other visual aids, and extensive testing of alternative versions of the scenario and payment vehicle prior to the conduct of the survey itself. It also pointed to conducting the survey by means of face-to-face interviews of respondents. Given previous experience suggesting that respondents have difficulty with WTA questions, requirement (b) was taken to imply that the question asked should be a WTP question. Plausibility was also taken, in part on the basis of the testing of alternatives in focus groups and the like, to require that the payment vehicle should be a one-off tax payment. The focus group work also assisted in designing an instrument that would be seen as neutral. In regard to (d), where scenario construction necessitated choices between options, for which neither theory nor survey research practice gave a strong ranking, the option was chosen which would, it was thought, if it had any effect, produce a lower WTP. Thus, for example, respondents were not shown pictures of oiled birds.

The development of the survey instrument used took place over a period of 18 months, and involved initially focus groups, followed by trial interviews and pilot surveys. The form that it finally took was as follows. After being asked about their views on various kinds of public goods and knowledge of the *Exxon Valdez* incident, respondents were presented with information about Prince William Sound, the port of Valdez, the spill and its environmental effects, and a programme to prevent damage from another spill. The programme would involve two coastguard vessels escorting each loaded tanker on its passage through Prince William Sound. These vessels would have two functions: first, reducing the likelihood of a grounding or collision, and second, should an accident occur, keeping the spill from spreading beyond the tanker. The interviewer then stated that the programme would be funded by a one-off tax on oil companies using the port of Valdez and that all households would also pay a one-off tax levy. Before asking about willingness to pay this tax, the interviewer presented material about the reasons why a respondent might not want to pay such a tax, so as to make it clear that a 'no' vote was socially acceptable.

The WTP question was whether the respondent would vote for the programme, given that the one-off household tax would be an amount $\$x$. The survey involved four different treatments in which the amount x varied as shown in Table 12.3 in the column headed A-15, which was the first WTP question number in the survey instrument. Depending on the answer to that question, a second WTP question was put to the interviewee. If the A-15 answer was 'yes', the respondent was asked whether he or she would vote for the programme if the tax cost were to be the higher amount shown in the column headed A-16. If the answer at A-15 was 'no', the interviewee was asked about voting given a tax cost at the lower amount shown in the column headed A-17.

Table 12.3 Monetary values used in WTP questionnaire for various treatments and various questions

Treatment	A-15	A-16	A-17
A	10	30	5
B	30	60	10
C	60	120	30
D	120	250	60

After the WTP questions, the interviewer asked a number of debriefing-type questions about the motives for the responses given, about attitudes and beliefs relevant to the scenario, and about the respondent's demographic and socio-economic characteristics.

The survey was conducted using a stratified random sample of dwelling units in the USA. Approximately 1600 units were selected. Given the cost that producing and using foreign-language versions of the survey instrument would have involved, non-English-speaking households were dropped from the sample. Within the remaining households, one respondent was randomly selected. Respondents were randomly assigned to one of the four WTP treatments. The response rate, based on sample size after dropping non-English-speaking households, was 75.2%.

The second column of Table 12.4 gives the proportion of 'yes' responses to the first WTP question, A-15, across the four treatments. The next four columns give the proportions for

Table 12.4 Response proportions

Treatment	Yes	Yes–Yes	Yes–No	No–Yes	No–No
A ($10, 30, 5)	67.42	45.08	22.35	3.03	29.55
B ($30, 60, 10)	51.69	26.04	26.04	11.32	36.60
C ($60, 120, 30)	50.59	21.26	29.13	9.84	39.76
D ($120, 250, 60)	34.24	13.62	20.62	11.67	54.09

Box 12.3 ***continued***

response patterns over the two WTP questions that all respondents were asked. Thus, for example, in the third column 45.08% of respondents asked initially about \$10 said 'yes' to it and to the \$30 that they were subsequently asked about, while in the fifth column 11.67% of the respondents initially asked about \$120 said 'no' to it but 'yes' to the \$60 that they were subsequently asked about. For the 'yes' answer to the first WTP question, and for the 'yes–yes' and 'no–no' patterns over the two questions, the entries in Table 12.4 look consistent with the basic idea that the probability of a 'yes' vote falls with the price tag attached. Note that the group answering 'no–no' may, as well as including respondents whose WTP lies between zero and the second \$*x* put to them, include respondents who do not think that the escort ship plan would work or think that, as a matter of principle, the oil shippers should bear the whole cost.

To use the response data to estimate a measure of average WTP, it is necessary to adopt some statistical model assumed to be generating the responses. In this study it was assumed that the underlying distribution of WTP was a Weibull distribution. Estimating the parameters of this distribution using maximum likelihood estimators and the response data gave an estimate of \$30.30 (95% confidence interval \$26.18–\$35.08) for the median WTP and of \$97.18 (95% confidence interval \$85.82–\$108.54) for mean WTP. In using the response data here, 'not sure' responses to either WTP question were treated as 'no' responses, consistent with the goal of producing a conservative estimate of average WTP.

A valuation function, using data on respondents' beliefs, attitudes and characteristics to construct explanatory variables for a regression with WTP as dependent variable, was estimated and the result was taken as demonstrating construct validity. The belief that in the absence of the escort ship programme the damage occurring in the future would be greater than in the *Exxon Valdez* case was positively associated with WTP. Again, it was found that a respondent's self-identification as an environmentalist was positively associated with WTP, other things being equal, as was an expectation of a future visit to Alaska. WTP was found to be positively associated with income.

Taking the estimated median WTP of \$30.30 as the relevant average and multiplying it by the number of English-speaking households in the USA gives a total WTP for the escort ship programme of \$2.75 billion. This was interpreted as representing an estimate of the lower bound on the correct, WTA-based, valuation of the TEV lost as a result of the *Exxon Valdez* oil spill.

Sensitivity analysis was conducted using the estimated valuation function. Thus, for example, the dummy variables representing beliefs about what the impact of a future spill would be in the absence of the escort vessels were all set to zero and the estimated function was then used to generate respondents' WTP. The intention here was to produce estimates of the WTP responses that would have arisen if all respondents had had the same belief in the efficacy of the escort vessel programme. The median of the individual WTP estimates so produced was \$27, to be compared with \$30.30 based on the actual responses. On the basis of several such experiments using the valuation function it was concluded that the result used at step (4) was reasonably robust.

Another type of sensitivity test involved re-running the survey. The survey described here was conducted in 1991. Two years later the same survey instrument was used again with a national sample and 'almost identical' results were obtained.

Source: Carson *et al.* (1995)

12.4 Choice experiments

The technique of choice experiments (CE) represents another type of stated preference survey-based valuation technique. CEs have now become a very popular way of valuing environmental goods and are also widely encountered in the marketing and transport economics literature. For interesting applications see Adamowicz *et al*. (1994), Boxall *et al*. (1996) and Birol *et al*. (2006).

CEs confront survey respondents with a number of discrete alternatives. These alternatives are described in some detail and individuals are then instructed to identify their most preferred alternative. In environmental economics applications the alternatives typically refer to competing environmental projects, whereas in transport economics, for example, they might refer to different modes of transport, and in marketing, the alternatives typically refer to different brands available for purchase. The alternatives are themselves fully described in terms of a finite number of attributes or, more precisely, by the level taken by these attributes.

One of the attributes is oftentimes the cost of choosing a given alternative and the ultimate objective of most CEs is to determine the trade-off between the levels of the remaining attributes against cost. For example, an application involving the choice between different modes of transport might include amongst other things journey time, the probability of getting a seat, presence of air conditioning and cost as relevant attributes. The interesting (and for transport planners clearly very important) question is how much people are WTP for shortened journey time.

There are several reasons underlying the growing popularity of CEs in environmental economics. Like CVs they can be used to measure non-use values and the control of the experimental design resides in the hands of the researcher. But at the same time CEs avoid the problem of yea-saying and nay-saying thought present in close-ended CVM surveys. Furthermore, in CEs monetary values are implicit rather than explicit and this may diminish the tendency of respondents to refuse to participate. In other words, at no point is the respondent asked how much they are WTP for a change in environmental quality. To illustrate, an individual might be prepared to participate in a CE relating to choice of transport mode in which the probability of a fatal accident is included as one of the modal attributes. But the same individual might not be willing or able to answer directly the question posed in a CV study, 'What is your WTP for a small reduction in the probability of a fatal accident?'

Perhaps their greatest advantage, however, lies in the fact that CEs make it possible to calculate WTP values for projects even if the attribute levels which define the project change. This point is explained more fully in the following section. WTP values derived from CEs can also be used in different contexts e.g. the value of commuting time derived from one study can be used to inform many different transport related infrastructure projects.

12.4.1 An example of choice modelling

We will now introduce the basic terminology and procedures underlying CEs in the environmental economics context by considering a fictitious example and drawing extensively on Bennett and Blamey (2001).

Imagine it is proposed that a forested area, currently subject to timber harvesting in parts and with uncontrolled recreational access in the remainder, becomes a bird sanctuary with no logging and restricted recreational access. An environmental CBA is commissioned in order to determine whether the project is welfare-improving. Basically, the question is whether the present value of aggregate WTP for the presumed environmental enhancement is equal to or greater than the opportunity costs of conservation in the form of forgone profits from logging and the diminished opportunities for recreation (notice that in this example there are both environmental benefits and environmental costs associated with the project).

The 'attributes' of the project are the number of bird species, the area of old-growth forest, the permitted number of visitors per year, and critically, the cost to the average taxpayer. Let us suppose that these attributes can take three different 'levels' displayed in Table 12.5

Selecting a value for each attribute generates an 'alternative'. For example, combining 5 bird species, 1800 hectares of old growth forest, 2000 visitors per year and a cost to taxpayers of \$10 constitutes one possible alternative. Grouping several different alternatives together forms a 'choice set' from which the individual is asked to choose their most preferred alternative. An example of a choice set containing three alternatives and accompanying rubric is displayed in Table 12.6.

In fact each respondent in a CE survey might be presented with several different choice sets each requiring the respondent to select his or her preferred alternative. Note that to be consistent with

Table 12.5 Project attribute levels

Bird species	5	10	15
Old growth forest (hectares)	1500	1800	2000
Visitors per year	4000	3000	2000
Cost per taxpayer (\$)	0	10	20

Table 12.6 an example of a choice set and accompanying rubric

Please consider carefully each of the following options. Given that only these options are available, which one would you choose?

Attribute	Alternative 1 Status Quo	Alternative 2	Alternative 3
Bird Species	5	15	10
Hectares Old Growth	1500	1800	1500
Annual visitors	2000	1000	2000
Cost to you ($)	0	20	10

Please circle your preferred option.

I would choose the status quo at no cost to me.........................1
I would choose alternative 2 at a cost to me of $20....................2
I would choose alternative 3 at a cost to me of $10......................3

welfare theory, each choice set must include the status quo as an alternative.

It will be instructive at this point to compare the foregoing CE with an alternative CV dichotomous-choice-based approach to valuing just one of the possible alternatives. The key CV question might run as follows: 'Suppose that the opportunity arose to increase the number of endangered species from 5 to 10 whilst simultaneously increasing the areas of old-growth forest from 1500 hectares to 1800 hectares and restricting visitor numbers to 2000 visits per annum. Would you be willing to pay a $X to secure such a change?' In this approach, consistent with the dichotomous choice approach, $X would be varied across the sample of respondents as $10, $20, $50 and $100 and the results analysed in the manner outlined earlier in this chapter.

While the CV approach would provide a WTP estimate for a change from the status quo to 10 endangered species, 1800 hectares and 2000 visits per annum, it would be impossible to then use the results to determine WTP for a change to, say, 16 endangered species with 2100 hectares of old-growth forest and a limit of 2500 visitors per annum. Instead, it would be necessary to conduct another CV experiment, which would be both time consuming and expensive. By contrast, the CE approach enables WTP values to be constructed for any environmental project because the CE technique derives the implicit value of project attributes. This flexibility enables decision makers to investigate alternative options without the need to commission further valuation studies.

12.4.2 Conducting a choice experiment

Conducting a CE involves many of the same steps involved in conducting a CV. This means identifying a target population whose values are to be measured and then developing the survey instrument using focus groups and a small-scale pilot survey. There are, however, a number of additional issues specific to CEs.

First of all, one has obviously to enumerate the attributes of the environmental project being valued. It is unusual to consider more than four or five attributes. Secondly, one has to select varying levels for those attributes. At least two different levels are required and, generally, attribute levels should span the range of interest to the policy maker commissioning the study. Attributes can take continuous values, e.g. the number of different species present, or alternatively discrete values, e.g. whether the water in the lake is suitable or not for swimming. Thirdly, one has to construct choice sets comprising the various alternatives. This is undoubtedly the most complicated stage of designing a CE and we will discuss this particular task at length.

The 'full factorial design' is the factorial combination of all possible attribute levels. In the example given above there are 3 attribute levels and 4 attributes. The full factorial design is therefore $3^4 = 81$ possible combinations of attribute levels. These could then be randomly combined into choice sets each comprising a finite number of alternatives (often three or four). As outlined, each choice set would have to include the status quo alternative.

But while a full factorial design offers certain advantages it is usually impractical to undertake a CE in which all chosen attribute levels have been included. Instead a 'fractional factorial design' is typically favoured, using only a subset of all possible combinations of attribute levels. Fortunately, statistical software packages such as SPSS contain routines to assist with the non-trivial task of determining what subset to include. SPSS will enquire about the number of attributes, the attribute levels, and the desired number of choice sets and then output an experimental design. Those without access to SPSS or equivalent software can simply copy the experimental designs used by other researchers.

Randomly sampling from fractional or full factorial designs in order to create choice sets may result in dominated alternatives and implausible attribute combinations. Dominated alternatives occur when in one choice set there exists an alternative that no individual would rationally select because it is in all respects inferior to another available alternative. Implausible attribute combinations arise when good reason prevents two attribute levels from occurring together. Suppose for example that we are valuing an environmental project concerning a river, including as attributes river flow and the number of fish species. It is clear that allowing the river to run dry taken in conjunction with a positive number of fish species is an implausible combination. Choice sets containing dominated alternatives or implausible combinations of attribute levels are typically discarded lest the respondent starts to express doubts about the entire survey.

Lastly, one has to consider how many choice experiments to include in any one interview. A tension exists between on one hand wringing as much information as possible from each respondent given the cost of conducting an interview and on the other hand maintaining the interest and participation of the respondent without which the survey is likely to suffer deleterious consequences.

12.4.3 Analysing the data

Following completion of the main survey the resulting data have to be coded in preparation for econometric analysis. This involves recording the choices offered to each respondent in terms of the attribute levels as well as the selection ultimately made by the respondent. For each choice set there will typically be as many lines of data as alternatives in each choice set. For each line there will also be a binary variable indicating whether the choice was the preferred option.

The data that a CE survey generates are once more analysed using the Random Utility Model (RUM). This model assumes that respondents have consistently selected those alternatives conferring the highest level of utility. More specifically, if individual i chooses choice g out of $j = 1, \ldots, J$ alternatives it must be that:

$$u_{ig} \geq u_{ij} \ \forall j \neq g \tag{12.12}$$

where u is the utility associated with the choice. If we further assume that utility comprises a deterministic component v and a stochastic component ε then we can write:

$$v_{ig} + \varepsilon_{ig} \geq v_{ij} + \varepsilon_{ij} \forall j \neq g \tag{12.13}$$

The purpose of including the error term ε is to account for unobservable variations in taste. If it is also assumed that the error term follows the Gumbel distribution (also known as the Type I Extreme Value distribution) then the probability that individual i will select choice g out of the J alternatives is given by

$$\Pr(j = g) = \frac{\exp(v_{ig})}{\sum_{j=1}^{J} \exp(v_{ij})} \tag{12.14}$$

This model of probability corresponds to the Conditional Logit model of choice.[16] The likelihood function of the Conditional Logit model of choice is given by the following expression in which $i = 1, \ldots, N$ is the number of individuals and y_{ij} is an indicator variable which takes the value unity if a particular choice was chosen and is zero otherwise

$$LogL = \sum_{i=1}^{N} \sum_{j=1}^{J} y_{ij} Log\left[\frac{\exp(v_{ij})}{\sum_{j=1}^{J} \exp(v_{ij})}\right] \tag{12.15}$$

In the context of the example developed above, the deterministic component of the utility function of respondent i is most straightforwardly represented by a linear function of the attributes, with β_1–β_4 the parameters to be estimated

$$v_{ij} = \beta_1 COST_{ij} + \beta_2 BIRDS_{ij} + \beta_3 HECTARES_{ij} + \beta_4 VISITORS_{ij} \tag{12.16}$$

[16] At this point we note a continuing confusion in the literature. What some refer to as the Conditional Logit model of choice others refer to as the Multinomial Logit model. One model is used when attributes of the choices vary (corresponding to the case in hand) and the other model when the choices are purely a function of individuals' characteristics, for example the probability that a person votes for one of three political parties (Conservative, Labour or Liberal Democrat) as a function of individual characteristics (age, educational attainment, race, household income).

The marginal willingness to pay (MWTP) for attributes, sometimes known as the 'implicit value' or 'part-worth', can be calculated by the negative of the ratio of the coefficient on the attribute divided by the coefficient on the cost attribute (which represents the marginal utility of money). In the current example the MWTP for an additional hectare of old-growth forest is therefore given by

$$MWTP = -\frac{\beta_3}{\beta_1} \tag{12.17}$$

The WTP for a specific policy package is given by the utility of the package (v^1) minus the utility of the status quo (v^0) divided by the coefficient on the cost variable (note that the value of the monetary attribute is assigned a zero value at this stage). It is in this way that the economic value of a variety of possible policy alternatives can be evaluated.

$$WTP = -\frac{v^1 - v^0}{\beta_1} \tag{12.18}$$

The socio-economic characteristics of respondents can affect the MWTP for the attributes but these characteristics have to be interacted with the attributes. Models combining choice-specific and individual-specific data are referred to as the Mixed Multinomial Logit model.

The MWTP can also be affected by the level of another attribute, although in many studies only the 'main effects' are investigated. Once more to employ an example from transport economics, it seems plausible to suggest that the speed of a metro train journey and the probability of getting a seat are both important to the traveller. But the faster the journey the less important it is to get a seat. In this instance an interactive effect between two attributes is present.

Sometimes CEs are conducted in which attributes are allowed to take only two (extreme) values. Although dramatically reducing the number of possible attribute combinations, this prevents analysts from exploring potential non-linearities and two-way interactions. Indeed, even when attribute levels are allowed to take more than two values, few CEs seem to allow for non-linearities. In the current example it is plausible to assume that the MWTP for an additional hectare of old-growth forest varies depending on whether 5000 or 20 000 hectares are left untouched.

12.4.4 Some problems and extensions

The most widely recognised drawback of CEs compared to CV is that the former place greater strain on respondents' cognitive abilities. The danger is that respondents then resort to simple rules of thumb as a way of 'solving' the CE puzzle confronting them. For example, individuals might choose on the basis of just one attribute. Someone who chooses on the basis of only one attribute is said to display lexicographic preferences. Respondents might even select one alternative over another regardless of its attributes. This frequently happens when a name or label naturally appends to a particular alternative, e.g. 'the status quo alternative' or 'the environmentally friendly choice'. In fact the label probably conveys additional information to the respondent and in order to avoid confounding it is necessary to include up to J–1 alternative specific constants (ASCs) in models. This is usually (although not invariably) unnecessary in 'generic' models in which the options are labelled merely A, B, C or 1, 2, 3. If we felt that simply labelling alternative 1 the 'Status Quo' alternative in the example of the bird sanctuary caused some individuals to choose that option, we would write the deterministic component of the indirect utility function for that option in the following way

$$\begin{aligned} v_{i1} = {} & \beta_0 ASC_{i1} + \beta_1 COST_{i1} + \beta_2 BIRDS_{i1} \\ & + \beta_3 HECTARES_{i1} + \beta_4 VISITORS_{i1} \end{aligned} \tag{12.19}$$

while the indirect utility function for alternatives 2 and 3 would be as before

$$\begin{aligned} v_{i2} = {} & \beta_1 COST_{i2} + \beta_2 BIRDS_{i2} \\ & + \beta_3 HECTARES_{i2} + \beta_4 VISITORS_{i2} \end{aligned} \tag{12.20}$$

$$\begin{aligned} v_{i3} = {} & \beta_1 COST_{i3} + \beta_2 BIRDS_{i3} \\ & + \beta_3 HECTARES_{i3} + \beta_4 VISITORS_{i3} \end{aligned} \tag{12.21}$$

In this example ASC would take the value unity for the first (status quo) alternative. A positive value for the coefficient β_0 would indicate that respondents are indeed drawn to the status quo alternative for reasons other than those found in the attributes.

Asking individuals only about their most preferred option is potentially wasteful, especially in view of the cost of conducting interviews. If individuals can rank the alternatives giving their first, second and subsequent choices it is possible to conduct a Contingent Ranking (CR) experiment. A CR experiment is like a sequence of CEs with the first choice being made from *J* alternatives; the second from the remaining *J*–1 alternatives; the third from the remaining *J*–2 alternatives etc. CR experiments increase further the cognitive effort required of the respondent and the choices may not always include the status quo.

Box 12.4 Remnant vegetation and wetlands protection

Table 12.7 Example of the choice set for the Macquarie Marshes choice experiment

	Option 1 (Current situation)	Option 2	Option 3
Your water rates (one-off increase)	No change	\$20 increase	\$50 increase
Irrigation-related employment	4400 jobs	4350 jobs	4350 jobs
Wetlands area	1000 km^2	1250 km^2	1650 km^2
Waterbirds breeding	Every 4 years	Every 3 years	Every year
Protected species present	12 species	25 species	15 species

I would choose option 1 ☐
I would choose option 2 ☐
I would choose option 3 ☐
I would not choose any of these because I prefer more water to be allocated to irrigation ☐

Bennett *et al.* (2001) present an excellent example of the application of CE to the task of environmental valuation.

The Macquarie Marshes was once the largest area of wetland in New South Wales in Australia. But following the increased use of water for irrigation purposes the size of these wetlands has in recent years shrunk from 5000 to 1000 square kilometres. Consequently the frequency of water bird breeding-events has fallen from every year to once every four years, and the number of protected and endangered water bird species present has fallen from 34 to just 12.

A CE questionnaire was developed to value a project to allocate more water to the marshes by purchasing water rights from farmers. The purchase of water rights would be funded by a one-off tax on all households in New South Wales. A number of different projects were defined in terms of the following five attributes: water rates (RATES); irrigation-related employment (JOBS); wetlands area (AREA); the frequency of water bird breeding-events (BREED) and the number of protected and endangered species present (ENDSPECIES).

Over a single weekend researchers distributed 416 questionnaires throughout Sydney on a drop-off and pick-up basis, and eventually 318 usable questionnaires were obtained. Comparing the characteristics of the sample average with those of Sydney as a whole suggests that the sample is not wholly unrepresentative of the target population. An example of a choice set for the Macquarie Marshes CE is given in Table 12.7.

Table 12.8 Results from the conditional logit model

Attribute	Coefficient (Standard error)
ASC	–0.30* (0.19)
RATES (\$ per household)	–0.12E–1*** (0.81E–3)
JOBS (number)	0.17E–2*** (0.65E–3)
AREA (km^2)	0.56E–3*** (0.13E–3)
BREED (years between breeding)	–0.31*** (0.51E–1)
ENDSPECIES (number)	0.50E–1*** (0.97E–2)
Log Likelihood	–1756.947

* implies significance at the 10% level; ** implies significance at the 5% level; and *** implies significance at the 1% level of confidence.

Using the collected data the Conditional Logit model was estimated (Table 12.8).

Note first the inclusion of the alternative specific constant ASC. Somewhat contrary to convention, this variable takes the value zero when the current situation is selected and otherwise takes the value unity. Its negative value indicates that households have a

Box 12.4 *continued*

preference for the status quo (a common finding in CEs). The remaining variables are all statistically significant at the 1% level of confidence. As expected, the coefficients on RATE and BREED are negative whereas the coefficients on the remaining variables are positive. The implicit prices for the project attributes (corrected from the original paper) are given in Table 12.9.

These implicit prices may be used to calculate the WTP for a project that increases the area of wetland from 1000 to 1400 km^2, reduces the number of years between water birds breeding from 4 to 3 years, and increases the number of protected and endangered species from 12 up to 16. There is no effect on the number of jobs. Household WTP for such a change compared to the current situation is calculated in the following manner:

$$36.50 = -\frac{\begin{bmatrix} -0.30 + 0.0017 \times (4400 - 4400) \\ + 0.00056 \times (1400 - 1000) - 0.31 \\ \times (3 - 4) + 0.051 \times (16 - 12) \end{bmatrix}}{-0.012} \quad (12.22)$$

The WTP value of \$36.50 (corrected from the original paper) may seem small but remember that it is per household and that in order to derive societal WTP it should be multiplied by the number of households in Sydney (of which there were 1.5 million in 2001, according to the Australian Bureau of Statistics).

Table 12.9 The implicit values per household

Attribute	Implicit value per household (\$)
JOBS	\$0.14 per job
AREA	\$0.05 per km^2
BREED (years between breeding)	–\$25.83 per year between breeding
ENDSPECIES (number)	\$4.17 per protected species

12.5 The travel cost method

We now consider the first revealed-preference valuation technique. The travel cost (TC) method is used to value the recreational benefits of environmental resources such as forests, national parks, wildlife reserves, and sites offering fishing and hunting opportunities. Although access to these places is typically free, users nonetheless pay a ‘price’ in terms of the travel cost incurred by visiting the site. The TC method assumes that individuals react to changes in travel costs in the same way they would react to changes in an admission fee. And the fact that individuals reside at differing distances from the recreational site means that in effect they face differing ‘prices’ and hence ‘purchase’ differing numbers of trips in any given time period.

The TC model is credited to Hotelling who, in a letter to the US National Park Service dated 1947, suggested that the benefits of public lands might be estimated by defining a series of concentric zones around each park and recording the number of visitors from each zone. He correctly argued that, along with data on travel costs, this would provide sufficient information to estimate the consumer surplus arising from the availability of the park. For a first application see Clawson (1959).

Note that TC is an application of ‘weak complementarity’ (see Appendix 12.1). The weak complementarity assumption is that if the site is too expensive and no trips are made, then changes in the condition and availability of the site do not affect utility. Of course, there might be instances in which individuals do care about the condition or availability of sites even when it is too expensive for them to visit. This is because many sites have non-use values that the TC technique, like other revealed-preference techniques, is incapable of measuring.

To anticipate what follows, an important distinction can be made between TC models that calculate the value of a recreational site and those that can be used to calculate the value of changes in the quality of the site (which is often the more policy-relevant endeavour).

12.5.1 The theoretical basis of travel cost

In the basic TC model the individual’s utility (u) is determined by consumption of a composite commodity (x), the number of visits made to a recreational site (r) along with the quality of the site (q).[17] The individual maximises

$$u(x, r, q) \tag{12.23}$$

subject to

$$m + wt_w = x + cr \tag{12.24}$$

Non-labour monetary income is m, w is the wage rate, t_w is time spent working and c is the round-trip financial cost of visiting the site. The individual's budget constraint highlights the fact that the consumption of the composite commodity, with its price normalised at unity, requires the individual to devote their time to labour. But time spent visiting the recreational site is time away from work so there is also a time constraint given by

$$t = t_w + t_r r \tag{12.25}$$

where t is total time available and t_r is time required for a return trip to the site. Substituting this expression into the budget constraint and rearranging yields

$$m + wt = x + (c + wt_r)r \tag{12.26}$$

Define p the 'price' of the trip as

$$p = c + wt_r \tag{12.27}$$

The first-order condition for optimality gives

$$\frac{\partial u}{\partial r} = \lambda p \tag{12.28}$$

The Lagrange Multiplier λ can be interpreted as the marginal utility of money. This implies that the individual will 'purchase' additional trips until the value of the last trip is equal to the price paid. The demand curve for the number of trips is given by

$$r = r(m, w, p, q) \tag{12.29}$$

Equation 12.29 is in principle estimable since, as noted, the variation in individuals' distances from the site guarantees that t_r and hence p varies across the sample. Site quality, on the other hand, does not vary across individuals or with distance from the site and will therefore be subsumed into the constant term. Integrating the demand curve between the limits corresponding to the current price (determined by the individual's distance from the site) and the 'choke' price (the price at which the number of trips is driven to zero) yields an estimate of the individuals' consumer surplus arising from the recreational value of the site. This model is readily extended to situations involving more than one site, where the demand for trips is determined, as one would expect, by the price and quality of all sites.

12.5.2 The Zonal Travel Cost model

The most basic empirical variant of the TC model is called the Zonal Travel Cost (ZTC) model. The easiest way to explain the ZTC model is by going through the steps involved.

1. Define a set of i concentric zones surrounding the recreational site of interest.
2. Collect information on the annual number of visitors from each zone by means of a survey.
3. Calculate the visitation rate by dividing the number of visits arising from each zone by the population of that zone.
4. Using a standard value per unit distance travelled and a standard value per unit of time, calculate the return trip travel cost from each zone.
5. Estimate a regression equation linking the visitation rate (v) to travel costs (c) e.g.

$$v_i = \alpha + \beta c_i \tag{12.30}$$

6. Use the equation to predict visitation rates with different hypothetical entrance fees e.g. starting with £10

$$\hat{v}_i = \alpha + \beta(c_i + 10) \tag{12.31}$$

7. Calculate total visitor numbers by multiplying the predicted visitation rate by the zonal population and then sum across all i zones yielding a point on a demand curve.
8. Employ the same procedure to evaluate the effect of imposing various other hypothetical admission charges, e.g. £20, £30, £40, £50 etc., to identify additional points on the demand curve.
9. The final step is to estimate the total economic benefit of the site by calculating the area under the demand curve.

[17] This analysis follows closely the model in Freeman (1993).

There is no particular reason for choosing a linear functional form for the relationship between visitation rates and travel costs. This relationship could equally well be modelled as a logarithmic relationship i.e.

$$\ln v_i = \alpha + \beta \ln c_i \tag{12.32}$$

Choosing a different functional form can sometimes have significant implications for the estimate of consumer surplus.[18] Studies must therefore consider different functional forms and attempt to identify the 'best' of these by reference to some statistical criterion. The TC method can be applied even when there is an admission charge for the site. In this case, the admission charge should be added on to the travel costs. The dependent variable in the equation describing the relationship between visitation rates and travel costs is necessarily non-negative.[19] For an example of the application of a ZTC model see Navrud and Mungatana (1994).

Box 12.5 An illustrative ZTC model example

Basic visitor data for a national park with no admission charge are as follows:

Zone	Visits	Population (thousands)	Round-trip distance (miles)
1	15 000	2 000	10
2	48 000	8 000	15
3	11 250	2 500	20
4	45 000	15 000	25
5	34 000	22 660	30

Distance is measured from the centre of the zone. Assume we know the total number of visits in the year from each zone and that the travel cost per mile is £1. The first step is to estimate the parameters of the trip-generating function:

$$V_i = \alpha + \beta C_i \tag{12.33}$$

where V_i is the visitation rate per thousand from zone i and C_i is travel costs from zone i. Using Ordinary Least Squares we get the estimated trip-generating equation as:

$$V_i = 10.5 - 0.3C_i \tag{12.34}$$

The next step is to use this equation to derive the relationship between the visitation rate and a hypothesised admission price P. The key assumption is that individuals would react to an admission charge in exactly the same way that they would react to higher travel costs.

$$V_i = 10.5 - 0.3(C_i + P) \tag{12.35}$$

We will consider P varying in steps of £5. For $P = £5$ we predict the visitation rate from each zone as well as total predicted visits by multiplying the visitation rate per thousand with the population of each zone measured in thousands:

Zone	$C_i + P$	V_i	$V_i \times$ Population (thousands)
1	15	6	12 000
2	20	4.5	36 000
3	25	3	7 500
4	30	1.5	22 500
5	35	0	0
Total			78 000

Proceeding in the same way for $P = £10$ and so on, we get the following data on price and total visits data for the surrogate demand function:

P (£)	Total visits
0	153 250
5	78 000
10	36 750
15	18 000
20	3 000
25	0

The third step is to obtain from this data an estimate of consumers' surplus for the year. Given that in fact $P = 0$, consumers' surplus is the total area under this demand function, which is:

$$\begin{gathered}[(153\,250 - 78\,000) \times 5 \times 0.5] + [78\,000 \times 5] \\ + \\ [(78\,000 - 36\,750) \times 5 \times 0.5] + [36\,750 \times 5] \\ + \\ [(36\,750 - 18\,000) \times 5 \times 0.5] + [18\,000 \times 5] \\ + \\ [(18\,000 - 3000) \times 5 \times 0.5] + [3000 \times 5] \\ + \\ [3000 \times 5 \times 0.5] \\ = £1\,061\,875.\end{gathered}$$

[18] Hanley (1989a) reports the results for MCS per visit for four different specifications of the trip-generating equation fitted to travel cost and visit data, using the zonal averaging approach, for a forest site. The range is from £0.32 to £15.13. Many of the TCM applications reported in the literature do not provide this kind of sensitivity analysis, simply reporting an MCS result for the chosen functional form.

[19] When the dependent variable is censored the correct estimator is the Tobit model.

12.5.3 Generic shortcomings of the travel cost model

The TC suffers from several problems, making it difficult to implement in a satisfactory manner.

As previously discussed, a properly specified demand curve should include, amongst other things, the price of all substitute recreational sites. However, implementing this solution requires first identifying which sites serve as substitutes. TC applications do not always deal adequately with the substitute sites issue (see Caulkins *et al*., 1986, and Smith and Karou, 1990).

A further problem concerns multi-purpose trips. Often visitors to recreational sites have more than one purpose in mind e.g. going shopping or visiting relatives. Is it then inappropriate to attribute the entire travel cost to the recreational experience obtained from the site? There seems no satisfactory solution to the problem of multi-purpose trips, other perhaps than to identify when they have occurred, perhaps by means of a dummy variable, or to estimate separate demand functions for those engaged in multi-purpose trips.

Perhaps the greatest problem, however, involves determining an appropriate value for the opportunity cost of travel time. The theoretical TC model assumes that neither work, nor travel time, confers disutility, and that individuals are free to choose the number of hours that they work. Together, these assumptions enable us to equate the value of travel time with the wage rate. But these assumptions are quite unrealistic. Equating the value of travel time with the wage rate is therefore wrong.[20]

Chevas *et al*. (1989) argue that time has a 'commodity value' as well as an opportunity cost in the form of forgone earnings. By commodity value they mean time that directly generates utility. They find the commodity value of time to be positive and suggest that this is netted out prior to calculating consumer surplus. Randall (1994) argues that the theoretically correct way to measure travel costs is according to the perceptions of those travelling. Most applications of the TC, however, involve the analyst calculating travel costs in accordance with some convention. Randall calls the results 'researcher-assigned travel cost estimates' and argues that when they are used 'the resulting travel costs and welfare estimates remain artefacts of the travel cost accounting and specification conventions selected for imposition'. Feather and Shaw (1999) suggest including both travel time and travel cost as separate arguments in the visitation rate regression equation. Normally, however, travel time and travel costs are highly if not perfectly collinear.[21]

12.5.4 The individual travel cost model

The individual travel cost (ITC) model collects information on individuals' use of a site over the past 12 months by interviewing them either on site or in the comfort of their own home. If individuals are interviewed on site then the sample of respondents will contain only those individuals for whom the number of visits is greater than or equal to one. If individuals are interviewed off-site then the sample will contain large numbers of individuals who did not visit the site at all in the past 12 months.

One advantage of the ITC model is that if the precise location of the individual is identified, the shortest distance to the site, and also to substitute sites, can be calculated with a high degree of accuracy using GIS software and a map of the road network. The ITC model also permits the analyst to investigate the socio-economic determinants of site use, such as household income, age and gender. The ITC model, however, continues to suffer from all the generic shortcomings associated with the TC technique identified in the preceding section.

Because the number of trips taken is a non-negative integer, the ITC model is estimated using a Poisson model rather than Ordinary Least Squares or the Tobit estimator. Obtaining measures of consumer surplus from the Poisson model is simple. The Probability Density Function for the Poisson distribution is

$$\Pr(r_i = n) = \frac{e^{-\lambda_i}\lambda_i^n}{n!} \qquad (12.36)$$

[20] Evidence from CEs suggests that individuals value their leisure travel time much less than their after-tax wage. A common convention is to see value travel time at one-third of the wage rate.

[21] See Garrod and Willis (1992) for evidence on the quantitative significance of the problem of measuring travel costs.

Where $n = \{0, 1, 2, \ldots\}$ and r is the number of trips made by individual i (we assume a survey undertaken off-site). Let the parameter λ be modelled as an exponential function of travel costs, c (other individual characteristics could also be included as covariates)

$$\lambda_i = e^{(\beta_0+\beta_1 c_i)} \tag{12.37}$$

Given that the expected value of the Poisson distribution is equal to the parameter λ the demand curve for the site is

$$r_i = e^{(\beta_0+\beta_1 c_i)} \tag{12.38}$$

To calculate the consumer surplus (CS) for a given individual, the demand curve is integrated between the limits of the current travel cost and infinity

$$CS_i = \int_{c_i}^{\infty} e^{\beta_0+\beta_1 c}\,dc \tag{12.39}$$

The definite integral is given by

$$CS_i = \left[\frac{e^{\beta_0+\beta_1 c}}{\beta_1}\right]_{c=c_i}^{c=\infty} \tag{12.40}$$

Given the expected negative value for β_1 the expression for CS is simply

$$CS_i = \frac{-r}{\beta_1} \tag{12.41}$$

Average CS per respondent would then be multiplied by the number of individuals living within the area surveyed to determine the site's current recreational value.[22]

Box 12.6 The value of recreational fishing in the Brazilian Pantanal

Shrestha *et al.* (2002) conduct an ITC analysis to estimate the value of recreational fishing in the Brazilian Pantanal. This is a region comprising 138 000 km^2 of wetlands in the centre of South America. Recreational fishing is an activity of growing economic importance to this area and between May 1994 and April 1995 46 000 anglers visited the region.

A survey was conducted of recreational anglers while they were weighing their catches at the mandatory weighing stations in Miranda and Corumba in Matto Grosso do Sul. Amongst other things, individuals were asked about the number of trips (TRIPS) taken in the past 12 months along with their round trip financial cost (TCOST) in US dollars and round-trip travel time (TTIME) in hours. Note that this study is unusual in that following the advice of Randall the authors rely on respondents' perceptions of these things and that following Feather and Shaw, both variables are included in the regression equation.

Out of 526 individuals interviewed 286 questionnaires were completed. The majority of those interviewed were Brazilian males who claimed to derive benefits not only from recreational fishing but also from the quality of the environment. Because the interviews were conducted on site the number of visits in the past 12 months is a strictly positive integer and the results displayed in Table 12.10 were obtained using a truncated Poisson model.[23]

Table 12.10 The regression results

Variable	Coefficient	Standard Error
TCOST	−0.00185	0.00047
TTIME	−0.0303	0.00916
CONSTANT	1.52	0.454
Pseudo R^2	0.57	

As expected, both TCOST and TTIME are negative and statistically significant. Furthermore, the use of perceived travel cost and perceived travel time enables, unusually, the authors to estimate the value of time. Comparing the coefficients on TCOST and TTIME the implied value of travel time is \$16.39 per hour.[24] Given the high (by Brazilian standards) average monthly income of \$4691, this seems not implausible.[25]

The CS per trip is given by the inverse of the coefficient on the travel cost variable. This evaluates to \$540.54 per trip or \$86.35 per day given the average trip length. Multiplying the number of recreational anglers by the mean number of trips gives 64 860 trips per year. The recreational value of the site to anglers is therefore \$35 059 424 per annum.

[22] The Poisson model is very simple but potentially it suffers from an econometric problem called 'over-dispersion' which more advanced techniques are able to overcome.

[23] Note that we have omitted from the table a large number of other explanatory variables. The truncated Poisson model adjusts for the fact that the number of trips is a strictly positive integer.

[24] This is obtained by dividing one coefficient by the other. Normally such a manoeuvre would not succeed since travel cost and travel time would be highly or even perfectly collinear, meaning that the value of time could not be estimated.

[25] Average gross annual income per worker in Brazil was \$9801 in 2003. The implied value of travel time is approximately half the gross wage rate.

12.5.5 The pooled travel cost model

Both the ZTC model and the ITC model consider the recreational value of one or more sites, but cannot value changes in the quality of sites. Quality variables for sites offering freshwater recreation might include angling success rates, water turbidity, surface area etc. Because the quality of the site is the same for all visitors, one cannot observe how changes in quality shift the demand curve. But many projects involve changes to the quality of a site rather than its complete destruction (which is what the ZTC and ITC models value).[26]

The Pooled Travel Cost (PTC) model can be used to estimate the value of changes in site quality, albeit at the expense of some very rigid assumptions. As its name would suggest, the PTC model combines travel cost data from different sites characterised by different levels of the quality attributes. Where r_{ij} is the number of trips to site j made by individual i, c is the travel cost and q is site quality, the PTC model assumes that

$$r_{ij} = \alpha + \beta c_{ij} + \gamma q_j + \varepsilon_{ij} \tag{12.42}$$

Sites with higher quality presumably generate more visits, other things being equal (the demand curve shifts out). The change in CS arising from a change in site quality is the difference in area between two shifting demand curves. The obvious weakness of the PTC model is that it neglects the existence of substitute sites and assumes that the number of individuals visiting any given site is unaffected by changes in the quality at any other site. These assumptions are unlikely to correspond to reality (see Smith and Desvouges, 1986).

12.5.6 Random utility models

Conventional TC models focus on the number of trips to a recreational site per time period but are unable to value the changes in site quality. The PTC model is able to estimate the value of changes in site quality but at the expense of unpalatable assumptions. The RUM, which we have already encountered in the context of CE models, considers choices made on particular choice-occasions but can readily value site characteristics and substitution between sites. The RUM approach is now ubiquitous in the travel cost literature.

For each choice-occasion the individual is modelled as selecting between a number of alternative sites J according to their travel cost and quality attributes. Thus individual i will choose to visit whatever site yields the highest utility v, although choices are also affected by an idiosyncratic error term ε. If the individual visits site $j = g$ then it must be that

$$v_{ij=g} + \varepsilon > v_{ij\neq g} + \varepsilon \tag{12.43}$$

If we once more assume that the error term follows a particular distribution (specifically a type 1 extreme value distribution) then the probability of the individual choosing a particular site labelled g can be modelled using the hopefully by now familiar Conditional Logit framework to give

$$\Pr_{ij=g} = \frac{e^{v_{ij=g}}}{\sum_{j=1}^{j=J} e^{v_{ij}}} \tag{12.44}$$

It is customary to adopt a linear functional form for the deterministic component of the indirect utility function where m is income, c is travel cost and q is site quality

$$v_{ij} = \beta_0(m_i - c_{ij}) + \beta_1 q_j \tag{12.45}$$

So that

$$\Pr_{ij=g} = \frac{e^{-\beta_0 c_{ij=g} + \beta_1 q_{j=g}}}{\sum_{j=1}^{j=J} e^{-\beta_0 c_{ij} + \beta_1 q_j}} \tag{12.46}$$

Note that the effect of adopting a linear representation of the indirect utility function is that m drops out of the expression. The negative of the marginal utility of income is the parameter on the cost of the trip, β_0. Parameter values are obtained via maximum likelihood techniques. RUMs can be used to

[26] One way of finding out about the value of changes in site quality is to ask individuals how their annual number of trips would change if the quality of the site changed. This is known as the contingent behaviour valuation approach (see Englin and Cameron, 1996 and Eiswerth *et al.*, 2000).

estimate the welfare changes associated with quality changes at one or many sites. Calculating a change in welfare following a change in site quality involves calculating the change in expected utility of the individual and dividing by the marginal utility of money. Expected utility is the utility of the various sites multiplied by the probability of actually visiting the site.[27] The CS for a change in site quality from q^0 to q^1 is given by the following expression

$$CS = \frac{\ln\left[\sum_{j=1}^{j=J} e^{v(q_j^1)}\right] - \ln\left[\sum_{j=1}^{j=J} e^{v(q_j^0)}\right]}{-\beta_0} \tag{12.47}$$

The RUM can also be used to calculate the CS for adding or deleting a site. The expression for adding a site is given by

$$CS = \frac{\ln\left[\sum_{j=1}^{j=J+1} e^{v(q_j^0)}\right] - \ln\left[\sum_{j=1}^{j=J} e^{v(q_j^0)}\right]}{-\beta_0} \tag{12.48}$$

It is possible to define welfare estimates for the loss of a particular site in analogous fashion. For a variety of applications see Hanley *et al*. (2003).

Turning now to weaknesses of the RUM, the total number of trips to a site is not explained but it is clear that this might change as a consequence of changes in site quality. The response is either to treat the number of trips as a separate problem or to expand the choice set to include choices over the number of visits to particular sites (see Parsons *et al*., 1999). The other slightly less obvious drawback of the RUM is the Independence of Irrelevant Alternatives (IIA) assumption. According to the Conditional Logit model, eliminating a site would cause visitors to redistribute themselves across the remaining sites in a way which leaves the relative probability of visiting those sites unchanged. This is often an unreasonable assumption, particularly when there is a close relationship between a subset of the sites included in the choice set.[28] It is possible to test for IIA using the test developed by Hausman and McFadden (1984). In recent applications the parameters of the RUM are treated as random variables allowing for preference heterogeneity (Train, 1998).

Box 12.7 Valuing deer-hunting ecosystem services

In a simple illustration of the RUM approach Knoche and Lupi (2007) value ecosystem services to deer hunting in the US state of Michigan. They argue that the establishment of particular types of ecosystems might provide recreational benefits in the form of hunting opportunities or the mere aesthetic appreciation of wildlife services. Cropland provides nutrition that can support increased deer populations compared to natural habitats. White-tailed deer populations are consequently higher in Southern Michigan where cropland comprises 39% of total land area. Nationally it is reported that 67% of farmers have experienced problems with deer and that the associated cost could be $100m annually. The State of Michigan currently leases agricultural land for hunting purposes, although the amount of land enrolled has declined due to diminished payments per hectare and farmers' concerns about hunting accident liability. An estimate of the benefits of accessible hunting areas might well be of interest to the public agency responsible.

The data used in the research were obtained from a survey of Michigan residents who had obtained a deer-hunting licence in 2002. Out of 2881 questionnaires, 1955 were returned. Amongst other things, hunters were asked about the number of trips taken to the site that they most regularly visited within a 50-mile radius of their home, and the number of trips to the most regularly visited site outside a 50-mile radius of their home. Accordingly, the two choice sets contain only those sites within a 50-mile radius

[27] Compare equation 12.47 with equation 12.18 used to value changes in the characteristics of particular recreational sites presented in the section on CEs. The difference arises because in the recreational RUM model of choice the individual has many alternative sites and which one is ultimately chosen is uncertain. In instances in which there are multiple choices available the welfare measure involves comparing the expected value of the *status quo* with the expected value of the scenario under investigation.

[28] Something called the 'nested' Conditional Logit model relaxes this assumption.

Box 12.7 *continued*

of their home and only those sites outwith a 50-mile radius of their homes. Trips taken during the firearm season and those taken in the archery season were separately identified. The sites were defined at the county level (there are 83 counties in the State of Michigan).

The authors calculate the distance from the respondent's home to the geographic centre of each county. The chosen price per mile was 5.2 cents per mile for gasoline and oil, 3.9 cents for maintenance, 1.5 cents for tyres, and 20.4 cents for vehicle depreciation. Insurance costs were excluded. The value for time was calculated by dividing the individuals' average income by 2080 (the number of working hours per year) and then dividing by three (it is a rule of thumb that the value of leisure time is very often taken to be one-third of the wage rate). Finally, to calculate time required for the trip an average speed of 40 miles per hour was assumed.

For each county the deer population DEER was obtained (in units of 10 000 deer). The amount of publicly accessible land ACCESS for the purposes of deer hunting was measured in terms of 100 000s of acres. Population of each county PEOPLE was included in units of 100 000s of persons and the area of each county SIZE was included in 1000s of square miles. Two dummy variables were included, denoting counties in the Upper Peninsula UP of the State of Michigan and counties in the Northern Lower Peninsula NLP (the Southern Lower Peninsula was the base).

The results of the Conditional Logit model are displayed in Table 12.11. Note that separate models are estimated for hunting with firearms and with a bow and arrow.

As anticipated, the coefficient on the travel cost variable PRICE is negative and statistically significant. As the cost of visiting a particular county increases the probability of visiting diminishes. Conversely, the population of deer and the amount of publicly accessible land increases the probability of a visit. The authors use the results to calculate the per trip and

Table 12.11 Results of the Conditional Logit model

Variable	Firearms Coefficient	Archery *P*-value	Coefficient	*P*-value
PRICE	−0.033	<0.001	−0.039	<0.001
DEER	0.130	<0.001	0.068	<0.001
ACCESS	0.056	<0.001	0.072	<0.001
PEOPLE	−0.102	<0.001	−0.063	<0.001
SIZE	−0.449	<0.001	−0.284	0.014
UP	2.329	<0.001	1.664	<0.001
NLP	0.490	<0.001	0.490	<0.001
No. Obs.	1416		685	
Pseudo-R^2	0.347		0.405	

Table 12.12 Welfare benefits of a 10 percent change in land accessible to hunters

	Per-trip benefit	Aggregate welfare
Firearm	$1.91	$ 8 754 000
Archery	$2.23	$10 250 000
Total		$19 004 000

aggregate benefits associated with a 10% increase in the amount of land accessible to hunters in the Southern Lower Peninsula area of the State of Michigan. These benefits are displayed in Table 12.12.

The aggregate welfare is calculated by multiplying the per-trip benefit by the average number of trips per hunter by the number of registered hunters in the State of Michigan (there were 5.6 trips per season per firearm hunter and 684 000 firearm hunters). Similar calculations are undertaken for aggregate benefits to hunters using a bow. Dividing total benefits by the number of additional acres implied by the 10% increase (an extra 480 000 acres) yields a benefit per acre of $39. This is much higher than the $5.55 per acre paid on average to farmers in Michigan's Hunter Access Program. The implication is that access to hunting land is an undersupplied good. This figure can also be compared to the per-acre value of agricultural produce of $443, suggesting that farmers might be able to increase their incomes significantly if they could capture these benefits.

12.6 Hedonic pricing

The hedonic price (HP) method is another widely used revealed-preference valuation technique. The HP method is usually, but not exclusively, applied to the property market within which many environmental goods are implicitly traded. Households reveal their preferences for these goods through their decision about where to locate. The HP method has been widely used to value household preferences

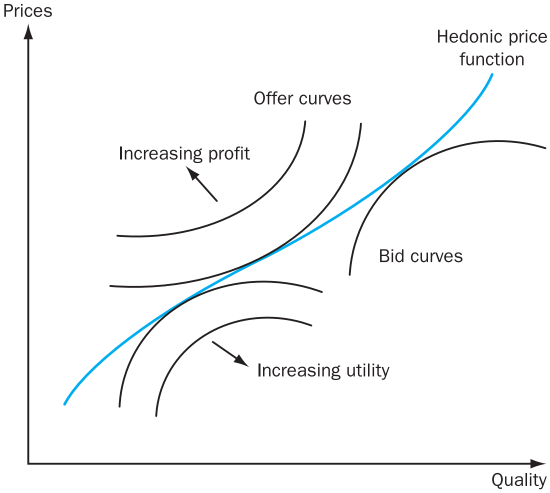


Figure 12.2 The hedonic price function as the double envelope of bid functions and offer curves

for noise nuisance, air quality, physical separation from locally undesirable land uses and the value of a statistical life.[29] The first application of the hedonic technique to valuing environmental goods is Ridker and Henning (1967). Interesting applications may be found in Garrod and Willis (1992) and Nimon and Beghin (1999).

12.6.1 Theoretical model

The first formal characterisation of the 'hedonic price function' was provided by Rosen (1974) building on Lancaster's characteristics theory of value (Lancaster, 1966). The hedonic price function describes the price of a quality-differentiated commodity in terms of its quality attributes. In fact the hedonic price function is the double envelope of buyers' bid functions and sellers' offer curves. The precise shape of the hedonic price function displayed in Figure 12.2 is determined by aggregate supply and demand for differing levels of the quality attributes within the market. Bid curves show all those combinations of prices and levels of the quality attribute that leave the buyer at the same level of utility. The slope of the bid function represents the maximum amount of money that the individual is WTP for an extra unit of the quality attribute. Offer curves show all those combinations of prices and levels of the quality attribute that leave sellers with the same profit.

To illustrate further the HP method we continue with the example of housing as a quality-differentiated good. Let h be the price of housing, $q_1, q_2, \ldots, q_n$ its characteristics and ε a random error term to represent the non-quantifiable aspects. The hedonic price function is thus written

$$h = h(q_1, q_2, \ldots, q_n) + \varepsilon \tag{12.49}$$

Taking the derivative of the hedonic price function with respect to the jth characteristic yields the implicit price function for that characteristic. Let p_j be the implicit price of that characteristic

$$p_j = \frac{\partial h(q_1, q_2, \ldots, q_n)}{\partial q_j} \tag{12.50}$$

Now let the household maximise its utility (u) which in turn depends upon the household's consumption of a composite good (x) and the level of the housing attributes (q). This maximisation problem is subject to a constraint linking the household's income (y), consumption of the composite good (whose price is normalised at unity) and the price of housing (h) which is, as we have already explained, a function of housing quality. The Lagrangian associated with the maximisation problem is

$$u(x, q_1, q_2, \ldots, q_n) + \lambda(y - x - h(q_1, q_2, \ldots, q_n)) \tag{12.51}$$

The first-order conditions indicate that the household's marginal willingness to pay (MWTP) for the environmental attribute is equal to the derivative of the hedonic price function evaluated at the household's chosen location.

$$\frac{\partial u/\partial q_j}{\lambda} = \frac{\partial h(q_1, q_2, \ldots, q_n)}{\partial q_j} = p_j \tag{12.52}$$

Conceptually, households can be thought of as moving along their demand curves for the environmental commodity until the unit price p_j of environmental quality is just equal to the MWTP. This is illustrated in Figure 12.3.

[29] The HP technique is an application of demand dependency. It is assumed that if the price of land in an area is prohibitively expensive then changes in the level of environmental quality have no value to the household.

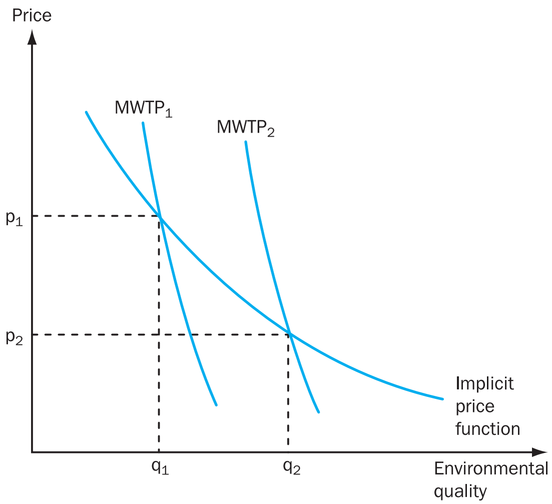


Figure 12.3 The implicit price function

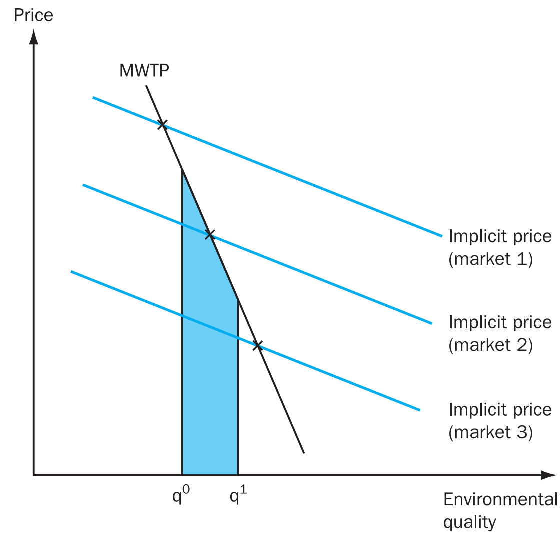


Figure 12.4 Identifying the MWTP curve

Evaluating this function at the attribute levels defining the household's chosen location yields the household's MWTP for this environmental characteristic.

Since we are frequently interested in evaluating non-marginal changes, there is a second stage to the HP technique that attempts to identify the household's demand curve for housing quality. The second stage involves relating the quantity of the *j*th characteristic consumed by household *i* to the implicit price of the amenities, the household's income, *y*, and other socio-economic characteristics of the household

$$q_{ji} = q_{ji}(p_{1i}, p_{2i}, \ldots, p_{ni}, y_i) \tag{12.53}$$

Early researchers erroneously believed that they could estimate this demand curve from the information provided by the implicit price functions taken from one market. In fact, the only way we can identify the demand curve is to have observations on households with the same socio-economic characteristics facing different implicit prices, such as might prevail in different cities or time periods as displayed in Figure 12.4. For an example of a study that attempts the second stage regression, see Day *et al*. (2007).

Obviously the demand for housing quality will differ between households. Poor households will be willing and able to pay less for housing quality than wealthier households. Note that we are ignoring the very real possibility that the price for characteristic *j* is not parametric to the individual but may vary depending on the quantity of the characteristic consumed by household *i* and the quantity of other characteristics consumed by household *i*.

Finally, integrating the demand curve between the different levels of quality *q* yields the benefit of improved quality to the household. This is likely to differ somewhat from the observed difference in house prices, especially for major changes in quality. Any improvement in housing quality, however, is likely to benefit landowners rather than tenants since improved quality will bid up the price of land. Widespread improvements in housing quality might also change the position of the hedonic house price function, complicating any attempt to estimate welfare changes.

12.6.2 Empirical implementation

The first step in conducting a hedonic house price analysis involves collecting information on the sale price of individual properties.[30]

[30] Early studies employed census tract data.

Whenever the dependent variable refers to sale prices rather than rental prices the MWTP for housing attributes corresponds to the present value of future benefits. And when sale prices are used rather than rental values the implicit price of environmental quality immediately reflects any anticipated change in environmental quality. For example, one would not expect to see a compensating price differential for a house situated in an area at risk of flooding if it had already been announced that a new flood relief project would soon entirely eliminate the risk of flooding. See McCluskey and Rausser (2003) for an example involving the discovery and remediation of toxic waste. One good reason for analysing house price data rather than data on rental values is that in some countries (e.g. Germany) rents are highly regulated and may not reflect market conditions.

Typically a hedonic house price study will have to control for the characteristics of property. This would be unnecessary if data on the price of bare land were available. In addition to house prices, therefore, the researcher also needs to record all relevant characteristics of the property itself e.g. the age of the property, type of property, length of the lease, presence of central heating, availability of off-street parking etc. Another important set of variables relates to accessibility e.g. distance to the nearest bus stop or train station, town centre, school, shopping centre etc. Also included would be the characteristics of the neighbourhood e.g. the rate of unemployment and the crime rate. Last but not least, one would include the environmental characteristics of the property such as night-time noise levels, ambient air quality, distance to the nearest landfill site etc.

Despite the inclusion of a large number of regressors, hedonic regressions are frequently able to explain only a small fraction of the variation in house prices, implying that important yet difficult to quantify characteristics have been omitted. Examples include the physical attractiveness of the building, the odour arising from nearby industrial activities and the quality of the view.

12.6.3 Underlying assumptions

Numerous assumptions underpin the HP method. The first is that there should be perfect information regarding the price and attributes of different properties. Without such information it is difficult to argue that the household has positioned itself at the point where household MWTP for every environmental amenity of interest is equal to the implicit price. But since a house is usually the single largest acquisition a household makes, it has every reason to become well informed prior to the purchase decision.

If the household is out of equilibrium the HP method assumes that it can without impediment move to another, more preferred location. However, if transaction costs are sufficiently high then they may outweigh the benefits of moving. Transaction costs in the property market are indeed substantial and include the time spent searching for properties, expenses on estate agents, lawyers and surveyors, stamp duty and removal costs.

It is necessary that the levels of all attributes vary continuously otherwise there might be a discontinuity in the implicit price function, and the location of the household might not reveal the household's true MWTP for that amenity.

Problems could arise if house price data are drawn from geographically or temporally distinct markets. Supply and demand conditions might differ between markets. One could potentially obtain very different hedonic price functions, as in Figure 12.5. Combining data from segmented markets is tantamount to

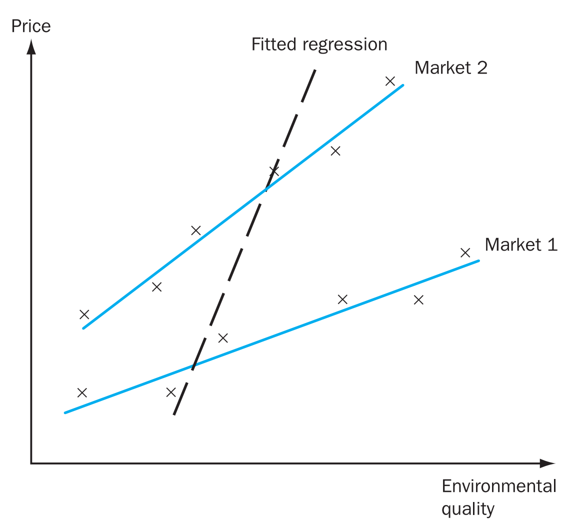


Figure 12.5 The effects of market segmentation

fitting a single regression to two or more spline functions. The resulting regression could yield biased estimates of the household's MWTP for key housing attributes. Analysts must therefore confirm the stability of the hedonic price regression (see Michaels and Smith, 1990). At the same, time market segmentation can be positively advantageous if it aids identification of household demand functions.

12.6.4 Econometric issues

Numerous researchers have attempted to measure households' preferences for air quality and the avoidance of noise pollution using the HP technique. Unfortunately, both air pollution and noise nuisance are generated by proximity to traffic. Consequently, it can be hard to attribute correctly the observed variation in house prices to one disamenity rather than the other. Equally, one should be suspicious of hedonic house price studies claiming to value marginal changes in noise nuisance while simultaneously omitting ambient air pollution from the hedonic price regression. The problem here is one of either multicollinearity or omitted variable bias.

A tension exists between the tractability of the functional form selected for the hedonic price equation and the ability of the regression equation to approximate the 'true' hedonic price function. It is common to see both the dependent and independent variables in the hedonic price regression transformed into logs and for interactive or quadratic terms to be included amongst the regressors (Cropper *et al*., 1988). And yet it would be a mistake to view the appropriate functional form as a purely empirical matter. For example, if the price of property was anything other than proportional to plot size this would create an incentive for property owners to 'repackage' their properties into larger or smaller units. In fact such repackaging is commonplace in the property market and researchers often select price per square metre as the dependent variable in their hedonic price regression on the assumption that any opportunities for repackaging have been exhausted.

Returning to an issue we flagged earlier, one of the implications of a non-linear functional form for the hedonic price function is that the implicit price of housing quality depends on the quantity consumed. Put another way, the prices of key housing attributes are no longer parametric to the individual. This causes the problem of endogeneity in the second-stage hedonic regression, leading to possible bias and inconsistency (see Taylor, 2003).

Lastly, as discussed, not all amenities of interest to households can be captured by the analyst and are instead consigned to the error term. Insofar as these omitted amenities are more similar for neighbouring properties, e.g. neighbouring properties share the same quality of view, then there is likely to be 'spatial autocorrelation' in the residuals. Researchers can obtain more efficient estimates of the parameters of the hedonic price function by adopting an estimator that anticipates the correlation between the residuals of observations corresponding to neighbouring properties, which is something that Ordinary Least Squares ignores (e.g. Kim *et al*., 2003).

Box 12.8 Valuing improvements in air quality in Los Angeles

Brookshire *et al*. (1982) took a sample of 634 sales of single-family homes which occurred between January 1977 and March 1978 in the Los Angeles metropolitan area. Data on two air pollution variables – nitrogen dioxide (NO_2) and total suspended particulates (TSP) – that are collected regularly at air monitoring stations in the area were used in the study. The objective of the study was to estimate property price differentials associated with air quality improvements for various localities within Los Angeles.

Housing sale prices were assumed to be a function of four sets of variables, H, N, A, and Q, where

H = housing structure variables (living area, number of bathrooms, etc.)
N = neighbourhood variables (crime rate, school quality, population density, etc.)
A = accessibility variables (distances to centres of employment, beaches, etc.)
Q = air quality variables (total suspended particulate matter and NO_2)

Box 12.8 ***continued***

Two hedonic price equations were estimated, one for each measure of pollution. Brookshire *et al.* searched through a variety of alternative functional forms for the hedonic equations, and those reported here are the ones that had the best statistical fit. In these two equations, note that the dependent variable is the natural logarithm of the home-sale price (in 1978 US$1000). Thus a change of one unit in any one of the explanatory variables results in a proportionate change of the dollar house-sale price, where the magnitude of that proportionate change is given by the estimated coefficient attached to the variable in question. However, in the cases where an explanatory variable also enters in log form, the associated coefficient gives the proportionate change in house-sale price that results from a unit proportionate change in the explanatory variable; i.e. it is an elasticity.

So, for example, if distance to the beach is increased by one unit (one unit is probably one mile although the paper does not define units), then the home-sale price will fall by 0.011586 in proportionate terms (by 1.1586%), if all other variables are held constant. A unit proportionate increase in NO_2 concentration (a 100% increase, or a doubling) results, *ceteris paribus*, in a proportionate decrease in house prices of 0.22407 (that is, 22.407%).

Approximately 90% of the variation in the home-sale price is accounted for by variation in the explanatory variables of the models (see the R^2 statistics). All coefficients have the expected sign and, except for those on crime, all are statistically significant at the 1% level. Particularly, the pollution variables have their expected negative influence on sale price and are highly significant. With the exception only of ethnic composition, the estimated coefficients on variables are very similar across the two reported equations.

Brookshire *et al.* use this information to calculate the rent premium that would be implied if air quality were to improve, for identical homes in given localities. These rent premia differ from one locality to another, but the results indicate rent differentials from $15.44 to $45.92 per month (in 1978 prices) for an improvement from 'poor' to 'fair' air quality, and from $33.17 to $128.46 (in 1978 prices) for an improvement from 'fair' to 'good' air quality. In each case, the higher figures are associated with higher-income communities.

Table 12.13 Results from the hedonic house price regression
Dependent variable = log (home sale price, in $1000)

Independent variable	NO_2 equation	*T*-statistics	TSP equation	*T*-statistics
Housing structure variables:				
Sale date	0.018591	(9.7577)	0.018654	(9.7727)
Age	−0.018171	(2.3385)	−0.021411	(2.8147)
Living area	0.00017568	(12.126)	0.00017507	(12.069)
Bathrooms	0.15602	(9.609)	0.15703	(9.6636)
Pool	0.058063	(4.6301)	0.058397	(4.6518)
Fireplaces	0.099577	(7.1705)	0.099927	(7.1866)
Neighbourhood variables:				
Log (Crime)	−0.08381	(1.5766)	−0.10401	(1.9974)
School quality	0.0019826	(3.9450)	0.001771	(3.5769)
Ethnic composition (per cent White)	0.027031	(4.3915)	0.043472	(6.2583)
Housing density	−0.000066926	(9.1277)	−0.000067613	(9.2359)
Public safety expenditures	0.00026192	(4.7602)	0.00026143	(4.7418)
Accessibility variables:				
Distance to beach	−0.011586	(7.8321)	−0.011612	(7.7822)
Distance to employment	−0.28514	(14.786)	−0.26232	(14.148)
Air pollution variables:				
log (TSP)			−0.22183	(3.8324)
log (NO_2)	−0.22407	(4.0324)		
Constant	2.2325	(2.9296)	1.0527	(1.4537)
R^2	0.89		0.89	

(Figures in brackets are *t*-statistic for the null hypothesis that the coefficient is zero.)
Source: Adapted from Brookshire *et al.* (1982)

12.6.5 Some extensions to the hedonic technique

The hedonic technique has been extended in a variety of ways, the most important of which are outlined below.

Some amenities differ in abundance within urban areas. Other amenities differ in abundance only between urban areas. In 'interurban' models the amenity value of environmental goods becomes capitalised into both land prices and wage rates. In this case, determining the household's MWTP for environmental amenities requires jointly estimating a hedonic house price function and a hedonic wage rate function. Let the household's maximisation problem be as before but now with the household obtaining wage income (w) that depends on environmental quality. The Lagrangian associated with this maximisation problem is

$$u(x, q) + \lambda(y - x - h(q) + w(q)) \tag{12.54}$$

The first-order conditions include

$$MWTP = \frac{\partial u/\partial q}{\lambda} = \frac{\partial h(q)}{\partial q} - \frac{\partial w(q)}{\partial q} \tag{12.55}$$

Note that the hedonic wage rate equation $w(q)$ explains wages as a function of the environmental characteristics of the location, the characteristics of the job and the characteristics of the worker. Note also that in interurban models it is possible that households living in an area with preferred environmental amenities might encounter higher house prices and (somewhat counter-intuitively) higher wages. This underlines the importance of considering both markets simultaneously (for an explanation see Roback, 1982). For an example of an interurban hedonic study see Rehdanz and Maddison, 2009).

The HP technique is also readily applicable to farmland, although in this context the HP technique is generally referred to as the 'Ricardian' technique. Such methods have been used to investigate the value to agriculture of topsoil depth, soil quality and climate.[31]

For every hectare the farmer is assumed to maximise net revenues π subject to a production function constraint. Note that the net revenue function is defined as the difference between crop revenues and all non-land costs. Production (x) is a function of the quantity of all non-land inputs (for the sake of simplicity here assumed to be only labour) as well as the environmental quality of the land, q. Let p be the price of output, w the price of labour and n the quantity of labour employed. The Lagrangian associated with the farmer's maximisation problem is then:

$$\pi = (px - wn) + \lambda(x - x(n, q)) \tag{12.56}$$

The result of solving the maximisation problem for n and substituting for x is a net revenue function which gives net revenues per hectare, net of farmland rents, as a function of output prices, non-land input prices and the environmental quality of the land

$$\pi = \pi(p, w, q) \tag{12.57}$$

Finally, the farmer maximises the difference between net revenues and the cost of renting a hectare of land h, which is itself a function of environmental quality. Differentiating this expression shows that equating the derivative of the net revenue function to the derivative of the hedonic price function is a first-order condition for optimality. Hence the value of a marginal change in the level of environmental quality is given by

$$\frac{\partial \pi}{\partial q} = \frac{\partial h(q)}{\partial q} \tag{12.58}$$

12.6.6 The hedonic technique and the value of statistical life

Many environmental projects involve small changes in the probability of premature mortality for unidentified members of the community. Sometimes changes in the risk of premature mortality will be the main objective of a project and in other cases any such changes will be purely incidental but nevertheless obviously an important consideration. Consider a scheme intended to cut emissions of harmful particulate matter in urban areas. In order to conduct an environmental CBA of such schemes a monetary value must be attached to the resulting change in the risk of premature mortality.

To some people the idea of attaching a monetary value to life is repugnant. However, it is not tenable to argue that the value of life is infinite and, accordingly, that all available resources should be devoted to improving safety irrespective of cost. It is evident

[31] For an example of the hedonic technique applied to agricultural land values see Mendelsohn *et al.* (1994).

that individuals regularly trade off cost and convenience against the risk of premature mortality in their everyday lives.

The value of statistical life (VOSL) is the willingness to pay to avoid a 1 in N chance of premature mortality aggregated over N individuals, and where N is a very large number. The VOSL indicates how an individual would behave when confronted by an infinitesimal change in the probability of their premature death. It does not mean that the individual would accept an amount equal to the VOSL in exchange for immediate death. The VOSL, moreover, is for the death of an anonymous individual.

Originally the VOSL was calculated from the expected lifetime earnings of individuals, although theory suggests that the VOSL should exceed expected lifetime earnings so such estimates provide only a lower bound (Bergstrom, 1982). The VOSL can also be determined using a variety of approaches to non-market valuation, most obviously CV. A sample of individuals might be asked about their WTP to reduce the risk of premature mortality by 1 in 10 000 and the average response multiplied by 10 000 to obtain the VOSL. Unsurprisingly perhaps, individuals seem to find it very difficult to answer questions involving (a) very small probabilities and (b) their own mortality. Fortunately, the hedonic technique can be extended to the labour market to determine the implicit value that individuals place on marginal changes in the risk of premature mortality. The technique takes advantage of the fact that the benefits and disbenefits of particular occupations, including the risk of work-related mortality, are capitalised into wage rates.

Empirical implementation of the hedonic wage technique entails analysing wages paid to a sample of workers engaged in different occupations across which the risk of a work-related fatality varies. We reiterate that the wage associated with each occupation is a function of both the characteristics of the worker and the characteristics of the job. This explains why occupations offering the highest remuneration often involve working in a comfortable office environment rather than down a coal mine or on a building site. The characteristics of the worker include their human capital proxied by years of education, years of work experience, ethnicity, age, and gender.

The key job characteristic is the occupation-specific risk of death through a work-related accident. Given that fatal accidents at work are rare, the risk of mortality should be estimated from multiple years of health and safety data. One ought also to include the risk of a serious yet non-fatal accident. Researchers, however, sometimes express the view that what matters more than the actuarial risk is workers' own perceptions of the risk of a fatal accident. Furthermore, some occupations present latent risks arising from exposure to chemicals in the workplace. Assuming that the periods over which remuneration is measured and the risk of death is calculated are identical, differentiating the hedonic wage function and evaluating the derivative at the chosen risk level of the individual worker reveals that worker's implicit MWTP for risk reduction (i.e. the VOSL).[32]

Box 12.9 The reward for risk in the labour market

Marin and Psacharopoulos (1982) present evidence on the VOSL in Britain using data from individual workers. The authors begin by defining a variable called GENRISK for each of 233 occupational groups, in the following way, with i indexing the occupational group

$$GENRISK_i = \frac{\textit{All-cause deaths}_i - \textit{Expected all-cause deaths}_i}{\textit{Number of employees}_i} \qquad (12.59)$$

The expected number of all-cause deaths is calculated allowing only for differences in age and social class. This measure of risk will, accordingly, include not only accidental deaths but also deaths arising from chronic impacts e.g. exposure to carcinogenic compounds in the workplace.

One of the several problems reported by the authors relates to the fact that individuals move between occupations or retire from the labour

[32] Despite the fact that the VOSL derived from hedonic wage studies has found widespread application in evaluating projects related to transport safety, there is nevertheless some doubt about whether the same values are applicable to evaluating schemes to improve air quality. Epidemiological evidence suggests that improvements in air quality will benefit mainly individuals who are over the age of 65 and already in a state of poor health. The willingness to pay of such individuals for reductions in the risk of premature mortality might differ from that of individuals young enough and fit enough to be in the workforce.

Box 12.9 *continued*

force when their health deteriorates. This explains why coal miners working underground have a lower risk of death than coal miners working above ground (miners start to work above ground when their health starts to deteriorate). Likewise, it is difficult to explain why publicans and innkeepers should have the third-highest excess risk of death other than by suggesting that some people are attracted into these professions because of the availability of alcohol and, consequently, die of cirrhosis or other alcohol-related diseases. For these and other reasons it can be expected that GENRISK is a somewhat inaccurate measure of the inherent riskiness of particular occupations.

The authors also employ an alternative measure of risk called ACCRISK which represents the actuarial risk of dying at an accident at work minus the expected rate given the occupational age structure. The authors argue that ACCRISK is more likely to represent perceived risks since, unlike the deaths caused by chronic ill health which GENRISK sought to include, deaths from accidents at work are more obvious and do not require detailed statistical analysis to unearth. Note that both measures of risk are subsequently expressed in terms of deaths per 1000 workers.

$$ACCRISK_i = \frac{\text{Accidental deaths}_i - \text{Expected accidental deaths}_i}{\text{Number of employees}_i} \tag{12.60}$$

The authors run an equation in which the logarithm of annual earnings (EARNINGS) is a function of years of schooling S, experience in the labour force EX, the percentage of workers covered by a collective agreement UNION, the number of weeks worked in the survey year WEEKS, a ranking of occupational desirability OCC, and one or other measure of risk. The UNION variable is included separately and interacted with one or other risk variable in view of the role that unions play in helping set the working conditions for their members.

The results displayed in Table 12.14 conform to prior expectation. In particular, the coefficients on the variables describing schooling, experience and union membership are positively signed and the coefficient on the variable describing the number of weeks worked in the survey year is approximately unity.[33] Most importantly, the coefficient on the risk variable is positive, indicating that labour markets do indeed compensate for an elevated risk of death.

Table 12.14 Results from the hedonic wage rate regression

	Parameter (T-statistic)	Parameter (T-statistic)
CONSTANT	1.9865 (26.09)	1.9433 (25.44)
S	0.0572 (23.63)	0.0583 (24.14)
EX	0.0466 (26.09)	0.0462 (34.42)
EX^2	−0.0008 (30.79)	−0.0008 (30.58)
Log WEEKS	1.1296 (61.43)	1.1307 (61.69)
GENRISK	0.0128 (2.10)	
ACCRISK		0.3663 (3.95)
UNION	0.0186 (7.58)	0.0020 (8.18)
OCC	0.0076 (21.44)	0.0081 (22.12)
UNION × GENRISK	−0.0004 (2.52)	
UNION × ACCRISK		−0.0046 (1.77)
R-squared	0.57	0.57
No. Obs.	5464	5509

The final task is to use these equations to derive the VOSL. For this purpose we will utilise the results in the final column of Table 12.14. Remembering that the dependent variable in the regression is the logarithm of earnings, the VOSL is given by following expression

$$\begin{aligned} VOSL &= 1000 \times \frac{\partial EARNINGS}{\partial ACCRISK} \\ &= 1000 \times (0.3663 - 0.0046 \times UNION) \times EARNINGS \\ &= £600{,}653 \end{aligned} \tag{12.61}$$

Note that the derivative is multiplied by 1000 because the risk is measured in terms of deaths per thousand. We have also evaluated the derivate at the sample average values for UNION (35.68) and EARNINGS (£2971). The estimate of £600 653 for the VOSL appears quite commensurate with other estimates published at the time, although it does, for reasons unknown to us, differ very slightly from the estimate provided by Marin and Psacharopoulos themselves.

[33] It would be odd if a 1% increase in the number of weeks worked resulted in anything other than a 1% increase in salary.

12.7 Production function-based techniques

So far we have valued the impact of changes in environmental quality on households. But environmental conditions are also of relevance to certain productive activities. Ozone, for example, is well known to have a deleterious effect on crop production. Although firms can adjust inputs of marketed commodities, they cannot alter ambient environmental quality.

We now consider valuation techniques in which the level of environmental quality is an argument in firms' production functions. We consider the welfare impact of both marginal and non-marginal changes in the level of environmental quality. The firm minimises wz where w is the price of inputs and z is the quantity of inputs. This is subject to the production function constraint

$$q(z, e) \geq q \tag{12.62}$$

where q is output and e is environmental quality. We assume that $\partial q/\partial e > 0$. This results in a series of conditional input demands for z. The optimised value of the function is the cost function $c(w, q, e)$. The contribution to social welfare arising from the production and consumption of the good is given by

$$W = \int_0^q p(q)dq - c(w, q, e) \tag{12.63}$$

Taking the derivative of this expression with respect to the level of environmental quality gives

$$\frac{\partial W}{\partial e} = \frac{\partial c(w, q, e)}{\partial e} \tag{12.64}$$

This says that the value of a marginal improvement in environmental quality is equal to the negative of the change in production costs. This is equivalent to multiplying the change in production by the price of output

$$\frac{\partial W}{\partial e} = p\frac{\partial q(z, e)}{\partial e} \tag{12.65}$$

In the case of non-marginal changes in the level of environmental quality the change in welfare is given by

$$\Delta W = \int_0^{q_1} p(q)dq - c(w, q_1, e_1) - \int_0^{q_0} p(q)dq + c(w, q_0, e_0) \tag{12.66}$$

This makes it clear that in order to compute the welfare impact of non-marginal changes it is necessary to know the effect of the change in production costs on the equilibrium level of output. In the equation e_0 is the initial level of environmental quality and e_1 is the final level of environmental quality. Similarly q_0 is initial output and q_1 is final output.

Box 12.10 The value of wetlands to the blue crab fishery of the Gulf Coast

The blue crab fishery on the Gulf Coast of Florida accounted for \$2.0 million of the \$68.1 million average value of total marine landings in Florida during 1972–1975. But this whole industry depends on the estuarine wetland areas of the same region which provide blue crabs, along with many other marine species, with spawning grounds, habitat, and food. The value of wetland acreage to this industry is the subject of a paper by Ellis and Fisher (1987).

Ellis and Fisher assume an iso-elastic demand function for crabs in which Q is quantity and P is price.

$$Q = KP^{-m} \tag{12.67}$$

They also assume a Cobb–Douglas production function

$$Q = AT^aL^b \tag{12.68}$$

where T is the effort in terms of setting traps and L is the area of wetland in acres. Note that unlike many papers Ellis and Fisher assume a 'static' model in which the size of the catch taken during one time period does not affect the relationship between catch and effort in any subsequent time period.

With W the unit cost of effort the industry cost function can be written as

$$C = WA^{-1/a}L^{-b/a}Q^{1/a} \tag{12.69}$$

Box 12.10 ***continued***

This makes the marginal cost of effort

$$\frac{\partial C}{\partial Q} = \frac{WA^{-1/a}L^{-b/a}Q^{(1-a)/a}}{a} \tag{12.70}$$

Any increase in wetland area leads to a downward shift in the marginal cost curve, resulting in an increase in total surplus. Along with the demand equation this system can be solved for P and Q for any given value of L. Using these algebraic results the change in total surplus can be readily calculated if the values of relevant parameters are known. Parameter estimates might be obtained using regression analysis on historical data or obtained through calibration.

In the Ellis and Fisher paper price is equated with marginal cost. Freeman (1991), however, notes that this would occur only in an optimally managed fishery. But there was no government regulation during the period covered by Ellis and Fisher. And in the absence of government regulation it seems safer to assume that the fishery was in fact characterised by open access.

If a fishery is open access then producer surplus is dissipated through uncontrolled increase in fishing effort. Effort expands until total revenue is equal to total costs

$$PQ = WA^{-1/a}L^{-b/a}Q^{1/a} \tag{12.71}$$

The system of equations can once more be solved for Q and P associated with any given amount L of wetland.

In either case, the value of the total surplus is given by the change in consumer surplus minus the change in costs where the L_0 and Q_0 refer to the original amount of wetland and original catch respectively, and L_1 and Q_1 to the new amount of wetland and new catch

$$\int_0^{Q_1} P(Q)dQ - C(Q_1, L_1) - \int_0^{Q_0} P(Q)dQ + C(Q_0, L_0) \tag{12.72}$$

Freeman estimates the change in economic surplus associated with an increase in wetland area from 25 000 to 100 000 acres under various management regimes. These calculations use assumed values for the elasticity of demand of between 0.5 and 10 (Table 12.15).

It is interesting to note that the whether the increase in wetland area is worth more to the optimally managed fishery or the open access fishery seems to depend on the elasticity of demand.

Table 12.15 The value to the blue crab fishery of changes in wetland area

Elasticity	Optimal Management ($)	Open Access ($)
0.5	248 009	269 436
10	116 464	47 898

Summary

The use of environmental CBA requires monetary measures of the utility changes associated with changes in the provision of environmental goods. In considering how changes in the provision of such goods impacts on individuals' utilities it is usual to distinguish between use and non-use values. Use values can be estimated using revealed-preference methods, which exploit data on observed behaviour. The main techniques here are TC and the HP method. With regard to non-use values there is no observed behaviour that contains relevant information, so that stated-preference methods have to be used. These involve asking individuals about their WTP, or WTA compensation for a hypothetical change in the provision of an environmental good. The most widely used technique is CV, which is subject to a large number of potential biases. A more recently introduced stated-preference technique is CE. The PF approach can be used to value the welfare impact of changes in environmental quality on firms' productive activities.

Further reading

Environmental valuation has generated a very large literature in the past four decades, and provided a substantial proportion of the articles in the most prestigious environmental economics journal, the *Journal of Environmental Economics and Management*. Smith (2000) looks at the role of this journal in development of environmental valuation over the quarter-century that the journal has existed. *Land Economics, Environmental and Resource Economics* and *Ecological Economics* are other journals where papers on environmental valuation appear with regularity.

By far the most comprehensive textbook on environmental valuation is Freeman (2003). Students may also consult Braden and Kolstad (1991) and Champ *et al*. (2003). Haab and McConnell (2002) deal with the econometric analysis of revealed-preference and stated-preference valuation data. Krutilla (1974) is a landmark paper.

Thorough discussions of the CV technique can be found in Mitchell and Carson (1989), Bateman and Willis (1999), Kriström (1999) and Bateman *et al*. (2002). The latter publication in particular offers a highly detailed, up-to-date explanation of how to carry out economic valuation using stated-preference techniques. See Alberini and Kahn (2006) for a collection of case studies. Hausman (1993) expresses anxieties about the use of CV to measure non-use values that continue to haunt researchers to this day.

CEs are described in meticulous detail by Louviere *et al*. (2000) and also by Hensher *et al*. (2005). For a survey of the issues involved in using CEs to value environmental goods see Hanley *et al*. (2001).

For a manual on how to use the TC technique to value nature see Ward and Beal (2000). For recent applications of the travel cost method see Hanley *et al*. (2003). Bockstael (1995) is a survey concentrating mainly on theoretical developments. See also Kling and Crooker (1999) for a review of models of recreational demand.

A review of the HP technique is contained in Nelson (1982). See also Palmquist (1991) and Palmquist (2003). For a meta-analysis of the impacts of air pollution on house prices see Smith and Huang (1995). To read more on the VOSL see Viscusi (1995).

The production function approach to valuing ecosystem services is well explained in Barbier (2007) and Hanley and Barbier (2009).

The technique of weak complementarity and weak substitutability is best analysed in Feenberg and Mills (1980).

Discussion questions

1. What different values might people have for stocks of the African Elephant?
2. Why do WTP and WTA differ and when is it appropriate to employ one rather than another?
3. Are CV studies unreliable?
4. Why have CE models become so popular in recent years?
5. Explain how you would conduct a travel cost study to determine the recreational value of Snowdonia national park.
6. Explain the problems involved in using the hedonic technique to value (a) non-marginal changes in the level of an environmental amenity and (b) the value of an environmental amenity whose level differs between cities.

Problems

1. Suppose an individual has the following utility function, where U denotes total utility and Q the quantity of a good or service consumed in a given period of time:

$$U(Q) = \alpha Q + \frac{\beta Q^2}{2}$$

   (a) Obtain the individual's marginal utility function.
   Assume $\alpha = 10$ and $\beta = -1/2$, and that the individual's consumption rises from Q_1 to Q_2, where $Q_1 = 2$ and $Q_2 = 4$.
   (b) What is the individual's marginal utility at Q_1 and Q_2?
   (c) Show that total utility can be interpreted as an area under an appropriate marginal utility function, and use this result to obtain the increase in total utility when consumption rises from Q_1 to Q_2.

2. Suppose that an individual has the utility function

$$U = E^{0.25} + Y^{0.75}$$

   where E is some index of environmental quality and Y is income. From an initial situation where $E = 1$ and $Y = 100$, calculate CS and ES for an increase in E to the level 2, and for a decrease in E to the level 0.5.

3. With E as some index of environmental quality and C_1 and C_2 as two 'ordinary' commodities, consider the following utility functions in regard to whether C_1 is non-essential and whether C_1 and E are weak complements:

   (a) $U = E^\alpha + C_1^\beta + C_2^\delta$
   (b) $U = E^\alpha C_1^\beta C_2^\delta$
   (c) $U = E^\alpha C_1^\beta + C_2^\delta$

4. Theory suggests that $\text{WTA}(y) = \text{WTP}(y + \text{WTA}(y))$ where y is income. This insight provides an opportunity to test the validity of the contingent valuation method. For example, Hammack and Brown (1974) surveyed the WTP of hunters to preserve duck populations estimating the following equation:

$$\ln(\text{WTP}) = 2.48 + 0.466 \ln(y)$$

   Suppose that individual preferences are given by the following indirect utility function:

$$V(P, y, z)$$

   where P is prices and z is environmental quality.

   (a) Use the indirect utility function to write down an expression for WTP for an increase in environmental quality from z^0 to z^1.
   (b) Now use the same function to write down an expression for an individual's WTA for a reduction in environmental quality from z^1 to z^0.
   (c) Write down the expression for WTP for an increase in environmental quality from z^0 to z^1 when the consumer's income is $y + \text{WTA}(y)$. Comparing this expression to the one given in 2, demonstrate that $\text{WTA}(y) = \text{WTP}(y + \text{WTA}(y))$ must indeed be the case.
   (d) In Hammack and Brown's paper, what is the WTP of a hunter with an income of \$12 100? Hammack and Brown predict a WTA of \$1044. Does such a high value for WTA refute or confirm the theory underlying contingent valuation?

CHAPTER 13 Irreversibility, risk and uncertainty

To know one's ignorance is the best part of knowledge.

Lao Tzu, The Tao, no. 71

Learning objectives

In this chapter you will

- learn about the difference between risk and uncertainty
- find out how risk affects environmental decision making and have the concepts of option value and option price explained
- see how irreversibility affects environmental decision making and learn about quasi-option value
- consider decision making in the face of uncertainty
- be introduced to the safe minimum standard and the precautionary principle
- learn how environmental performance bonds could work

Introduction

Much of our analysis has assumed that the consequences of decisions are known with certainty and are reversible. However, many of our discussions have implied that these assumptions are not factually correct. Resource decisions concern the future as well as the present, and we cannot know the future with certainty. Many such decisions have consequences that are irreversible. The ecological consequences of economic behaviour especially are frequently a matter of considerable ignorance, beyond the presumption of irreversible change. The assumption of certainty and reversibility in much of the literature is a simplifying device: it is convenient to assume away some real-world complexities in order to develop analytical insights. But simplifying things in this way is only appropriate when the thing being ignored does not have major consequences for the results of the analysis. It is important, therefore, to see what difference it makes to the analysis if it is assumed that the future is not known with certainty and may involve irreversible change.

The central objective of this chapter, then, is to consider how recognition of imperfect knowledge about the future and irreversibility affects resource and environmental economics. To orient our analysis, we shall consider the use of environmental cost–benefit analysis (ECBA) and particularly, to fix ideas, we will, as previously, locate the discussion in the context of a decision about whether to conserve a wilderness area or to allow it to be developed with the consequent loss of wilderness values. The insights developed in this context apply generally where there is incomplete knowledge and irreversibility.

The chapter is organised as follows. In the first section of the chapter we distinguish two kinds of imperfect knowledge, risk and uncertainty, and discuss individual behaviour in a risky world, leaving decision making in the face of uncertainty for later consideration. We then consider, in the second and third sections, option value and quasi-option value, which were mentioned, but not explained or discussed in the previous chapter. These arise when individual and social decisions have to be made in the face of risk. The fourth section draws on that discussion to consider ECBA which recognises risk. The penultimate section of the chapter discusses decision making in the face of uncertainty, and in the final section this analysis is used to consider the idea that, in the face of uncertainty combined with irreversibility, environmental policy should be cautious and adopt the safe minimum standard approach.

13.1 Individual decision making in the face of risk

In considering the implications of imperfect knowledge of the future, it is useful to distinguish between risk and uncertainty. Situations involving risk are those where the possible consequences of a decision can be completely enumerated, and probabilities assigned to each possibility. The possibilities are often referred to in the literature as 'states of the world' or 'states of nature', or just 'states'. Where the assignment of probabilities to all states is not possible, we are dealing with uncertainty. Two sorts of uncertainty can be distinguished. We mean by 'uncertainty' the situation where the possible consequences of a decision can be fully enumerated, but where the decision maker cannot assign probabilities. A more profound kind of uncertainty exists where the decision maker cannot enumerate all of the possible consequences of a decision – we call this radical uncertainty.

The distinction that we make between risk and uncertainty, originally due to Knight (1921), is not followed universally in the economics literature. Much modern usage conflates risk and uncertainty in Knight's sense under the general heading of uncertainty. So, for example, Freeman's definitive text on environmental valuation (Freeman, 1993, p. 220) uses the term 'individual uncertainty' to refer to 'situations in which an individual is uncertain as to which of two or more alternative states of nature will be realized'. However, in the context of environmental and resource economics, where some decisions must be made in the face of what can only be properly described as ignorance, we feel that it is useful to continue with Knight's distinction.

The classic risk situations are gambling and insurance. In the former case, unless cheating is involved, probabilities can be assigned to outcomes on the basis of the known properties of the gamble – as with betting on the toss of a coin or the spin of a roulette wheel. In insurance, probabilities are assigned on the basis of lots of past experience – as with life expectancies of individuals at different ages and in different circumstances, or with the incidence of accidents for motor vehicle drivers of different ages. In some gambling situations, such as horse racing, probabilities are also assigned on the basis of past 'form', albeit differently by different observers. Where there is no past 'form' and/or the underlying properties of the situation to be affected by the decision are not well understood, probabilities cannot be assigned by these means. This sort of situation is exemplified by the so-called greenhouse effect in relation to prospective climate change, discussed in Chapter 9.

In many environmental decision contexts probabilities are derived from models of the processes of interest. In the case of urban air pollution, for example, for given levels of emissions from a given set of sources, ambient pollution levels at locations will vary with meteorological conditions. Physical models of the airshed can be used to simulate probabilities of different ambient levels at locations of interest: see the discussion of ambient pollution levels in Chapters 5 and 7. Again, models of nuclear reactors have been used to calculate the probabilities of various kinds of accident, there being little 'form' to go on, and experimentation to establish actual empirical knowledge being out of the question.

Where probabilities are assigned on the basis of form or knowledge, they are sometimes referred to as 'objective' probabilities. Some economists deal with situations where the assignment of objective probabilities is seen as impossible by treating the decision-making problem as being dealt with by the assignment of 'subjective' probabilities. The idea is that the decision maker proceeds by assigning, on the basis of judgement, to each of the possible outcomes that he or she has identified a set of weights that satisfy the requirements for probabilities – basically they comprise positive numbers that sum to unity. However, this assumes that the decision maker feels able to do this, and, more fundamentally, feels able to enumerate all possible outcomes. In our view, it is more appropriate to admit that there are environmental decision-making problems, as exemplified by the greenhouse effect, which are not well characterised by these assumptions, and to consider uncertainty as distinct from risk. We defer discussion of decision making in the face of uncertainty until the final two sections of the chapter. Until then we proceed on the assumption that probabilities can somehow – possibly subjectively – be assigned to a complete enumeration of the outcomes considered possible.

13.1.1 The St Petersburg paradox

Consider the following potential gamble. A fair coin will be tossed repeatedly until it lands tail up. If it falls head up at the first toss, the gambler gets £1. If it falls head up at the second toss, the gambler gets £2, at the third toss £4, at the fourth £8, and so on. Tossing continues until the coin falls tail up. How much would somebody be willing to pay for such a gamble? The answer might appear to be 'an infinite amount' because the expected monetary value of the gamble is infinite. The expected value is the sum of the probability-weighted possible outcomes, which in this case is the infinite series

$$(0.5 \times 1) + (0.5^2 \times 2) + (0.5^3 \times 4) + (0.5^4 \times 8) + \dots$$
$$= 0.5 + 0.5 + 0.5 + 0.5 + \dots$$

which has an infinite sum. That anybody would be prepared to pay a very large amount of money for such a gamble violates everyday experience, and the example is known as the Bernoulli, or St Petersburg, paradox.[1]

The paradox can be resolved by assuming that individuals assess gambles in terms of expected utility, rather than expected monetary value, and that the utility function exhibits diminishing marginal utility. The relevant outcome is then the infinite series

$$0.5U(1) + 0.5^2U(2) + 0.5^3U(4) + 0.5^4U(8) + \dots$$

which has a finite sum, so long as there is some upper limit to U, which is what diminishing marginal utility implies. Diminishing marginal utility is a very natural assumption for economists. In economics, the basic approach to the analysis of individual behaviour in any kind of risky situation is to assume the maximisation of expected utility and diminishing marginal utility.

13.1.2 Basic concepts for risk analysis

The basic concepts used by economists here are expected value, expected utility, risk neutrality/aversion/preference, certainty equivalence and the cost of risk bearing. To develop these, consider an individual facing a gamble – though it could be any risky choice – where there are just two possible outcomes expressed in terms of the individual's income, Y_1 and Y_2. The probabilities associated with Y_1 and Y_2 are p_1 and p_2, where, by virtue of the fact that one of the outcomes must occur, $p_2 = (1 - p_1)$. Then, the expected value of the income outcome of the gamble is

$$\mathrm{E}[Y] = p_1Y_1 + (1 - p_1)Y_2 \tag{13.1}$$

where E[.] is the expected value operator. It says that we are referring to the expected value of whatever appears inside the square brackets. The term 'expectation' is sometimes used for 'expected value', so that equation 13.1 would be said to give the expectation of the gamble. The expected utility of the gamble is:

$$\mathrm{E}[U] = p_1U(Y_1) + (1 - p_1)U(Y_2) \tag{13.2}$$

If the utility function is given the algebraic form $U = Y^a$ where a is a positive fraction so that $\partial U/\partial Y > 0$ and $\partial^2U/\partial Y^2 < 0$, this is

$$\mathrm{E}[U] = p_1Y_1^a + (1 - p_1)Y_2^a \tag{13.3}$$

The certainty equivalent to this gamble is the Y corresponding to its expected utility; that is, the result of solving

$$U(Y) = \mathrm{E}[U]$$

for Y. For our case with $U(Y) = Y^a$ this is

[1] This paradox was posed by Bernoulli in the eighteenth century, and is sometimes known by his name. The origin of the name for the paradox used in the section heading here lies in the Bernoulli family's long association with St Petersburg.

$$Y^a = p_1 Y_1^a + (1 - p_1) Y_2^a \tag{13.4}$$

to be solved for Y, given p_1, Y_1, Y_2 and a.

Now consider Figure 13.1 for this gamble. Y^{**} is the expected value of the gamble. The straight line ACB is the locus of expected value/expected utility combinations for a gamble with just two outcomes Y_1 and Y_2 as p_1 varies. If $p_1 = 1$, so that $p_2 = 0$ and Y_1 is certain, using equations 13.1 and 13.2 we get point A with Y_1 and $U(Y_1)$. If $p_1 = 0$, we get B with Y_2 and $U(Y_2)$. If $p_1 = 0.5$, we get Y^{**} halfway between Y_1 and Y_2, E[U] equal to the vertical distance Y^{**}C. To the left of C along CA $p_1 > 0.5$, to the right along CB $p_1 < 0.5$.

The utility function maps certain income into utility. If $Y^{**} = \mathrm{E}[Y]$ were certain income, rather than the expected value of a gamble, the utility level corresponding would be that at point E on the $U(Y)$ curve, with $U(\mathrm{E}[Y])$ corresponding. The horizontal line C to E[U] cuts the $U(Y)$ curve at D, which corresponds to an income of Y^*. This is the certainty equivalent for this gamble, the solution for Y in equation 13.4, as it is the certain level of income that yields the same utility as the expected utility of the gamble.

Y^* is, in Figure 13.1 and generally for $U = Y^a$ with $0 < a < 1$, less than Y^{**}. The certainty equivalent is less than the expected value of the gamble. Put another way, the utility of the certain payment of Y^{**} is greater than the utility of a gamble with expected value Y^{**}. If this individual were offered the sum of money Y^{**} or a free ticket to the gamble described here, he or she would not be indifferent but would prefer the sum of money over the actuarially equal gamble. We say that such an individual is risk-averse. If in $U = Y^a$, a took the value 1 then the graph for $U(Y)$ in Figure 13.1 would be a straight line with ADEB coinciding with ACB, and the individual would be indifferent between the money sum and the free ticket; in other words, risk-neutral. If an individual had a utility function such that $\partial U/\partial Y > 0$ and $\partial^2 U/\partial Y^2 > 0$, instead of $\partial U/\partial Y > 0$ and $\partial^2 U/\partial Y^2 < 0$, then in a diagram like Figure 13.1 the arc ADEB would lie below the straight line ACB and the ticket to gamble would be preferred to the sum of money. Such an individual would be said to exhibit risk-preference.

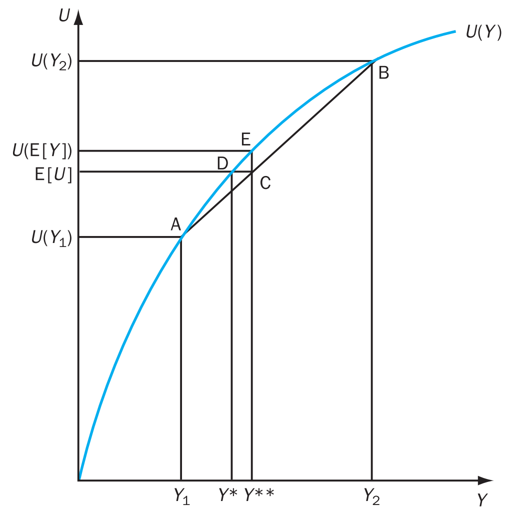


Figure 13.1 Risk aversion and the cost of risk bearing

Reflecting everyday experience, in economics it is assumed that the typical individual is risk-averse, as depicted in Figure 13.1. For such individuals, taking a risk is costly in utility terms, which cost can be expressed in a monetary measure using the concepts developed here. The cost of risk bearing, CORB, is defined as the difference between the expected value of the gamble and its certainty equivalent:

$$\mathrm{CORB} = Y^{**} - Y^* \tag{13.5}$$

CORB is analogous to the measures of surplus and variation developed in the previous chapter, in that it is a monetary measure of a utility difference, which in this case would arise, for a risk-averse individual, from being in a risky as opposed to an actuarially equivalent certain situation.

While we have developed these concepts here for the case of a gamble with just two equiprobable outcomes, they are not restricted to such a context, which was adopted solely for expositional convenience. The number of possible outcomes does not have to be just two, nor do all possible outcomes have to have equal probabilities attached to them. The situation underlying the outcomes does not have to be a gamble as generally understood – it could, for example, be a choice about whether to insure or not, or climatic conditions affecting agricultural output.[2]

[2] Our treatment here of the economic analysis of individual behaviour in the face of risk has been neither rigorous nor comprehensive. For fuller accounts see, for example, Kreps (1990).

Or, as discussed in the next section, the basic ideas can be used to consider the situation of individuals who do not know for sure what they will demand in the future, or what its availability will be.

13.2 Option price and option value

We now return to the context of a wilderness area for which some development is proposed and under consideration. The basic idea of option value was introduced by Weisbrod (1964) in considering a national park and the prospect of its closure. Park closure is equivalent, from the point of view of use value, to development driving the value of the wilderness area's amenity services to zero. We will adopt the particular Weisbrod context here. Weisbrod saw that as well as a loss to current visitors, closure would entail a loss to potential future visitors. He argued that the benefit of keeping the park open would be understated by just measuring current consumer surplus for visitors, and that there should be added to that a measure of the benefit of future availability. He called this additional component of preservation benefit 'option value'.

Weisbrod's definition of option value, and the claim that it was a preservation benefit additional to consumer surplus, led to some controversy. Eventually, Cicchetti and Freeman (1971) established a set of definitions that proved generally acceptable, and appeared to support the basic thrust of Weisbrod's position, at least in so far as individuals are risk-averse. We now set out a simplified version of their analysis, using the concepts developed in the previous section.

13.2.1 Risky availability

Consider an individual and a national park wilderness area. In Figure 13.2, $U(\mathrm{A})$ is the level of utility that the individual attains for some given level of income, Y_{A}, if she wants to visit and the park is open. 'A' is for available. Using N for not available, $U(\mathrm{N})$ is the utility experienced if the individual wants to visit and the park has been closed. Given non-availability, how much would the individual be willing to pay for availability? The answer is the sum of money $Y_{\mathrm{A}} - Y_{\mathrm{N}}$, which would restore utility to the level $U(\mathrm{A})$.

Now, that question and the answer imply that the individual is either in a situation where access to the park is available, or in a situation where it is not. The idea of option value relates rather to a situation in which the individual does not know for sure whether future access will be available or not. Figure 13.2 deals with this situation along the lines set out for Figure 13.1 in relation to a gamble. Assign a probability of p_1 to the N situation and $1 - p_1$ to the A situation. Then the straight line NCA in Figure 13.2 is the locus of U/Y combinations as p_1 varies – at N p_1 is 1, at A it is zero. As before, Y^{**} is the expected value of the outcome for some given p_1, and Y^* is the certainty equivalent.

The sum of money $Y_{\mathrm{A}} - Y^{**}$ is the expected value of the individual's compensating surplus, E[CS]. For p_1 at 1, Y^{**} would coincide with Y_{N}, and willingness to pay for availability would be $Y_{\mathrm{A}} - Y_{\mathrm{N}}$. For $p_1 = 0$, Y^{**} would coincide with Y_{A} and CS would be zero. For situations where $0 < p_1 < 1$, the expected value of compensating surplus as willingness to pay is determined by the value for p_1. As Figure 13.2 is drawn, $p_1 = 0.5$ and Y^{**} is halfway between Y_{N} and Y_{A}.

The sum of money $Y_{\mathrm{A}} - Y^*$, where Y^* is the certainty equivalent for this 'gamble' on availability, is what is known as 'option price', OP, the maximum

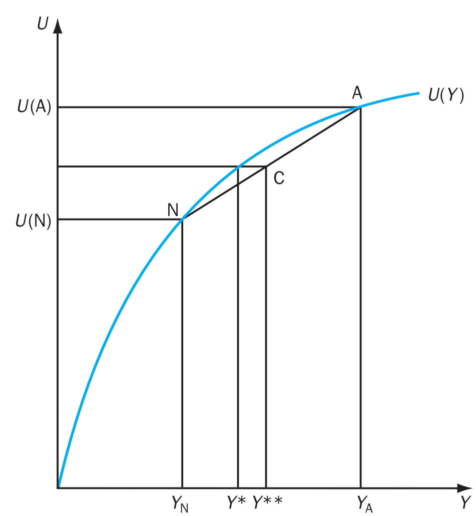


Figure 13.2 Risk aversion, option price and option value

amount that the individual would be willing to pay for an option which would guarantee access to an open park. As Figure 13.2 is drawn, $Y^{**} > Y^*$, so that OP is greater than E[CS]. Cicchetti and Freeman called the difference between OP and E[CS] 'option value', OV, with

$$OP = E[CS] + OV \tag{13.6}$$

with OV positive. From the previous section of this chapter we know that the way Figure 13.2 is drawn reflects the assumption of risk aversion. For a risk-neutral individual the straight line NCA would coincide with the arc NA, so that Y^* and Y^{**} would coincide, and OV would be zero with OP = E[CS]. As Cicchetti and Freeman put it, 'Option value is a risk aversion premium' (1971, p. 536). Weisbrod's idea was, in this framework, that E[CS] would understate the preservation benefits of keeping the park open, because risk-averse individuals are willing to pay a premium to avoid risk.

13.2.2 *Ex ante and ex post* measurement

Our report of the Cicchetti and Freeman analysis, in the interest of getting at the basic idea, was not entirely accurate. In particular, we treated risk as attaching to availability where the individual knows that she will want to visit in the future, whereas in the original formulation it (also) attaches to the individual's future preferences in an analysis of the policy decision as to whether to allow development to close the park or keep it open. We now explore option value further in that context, distinguishing between *ex ante* and *ex post* perspectives.[3] The *ex ante* view is prior to outcomes being revealed; the *ex post* is after the event, when the outcomes are known.

We need now to introduce some additional notation. We will use s_k to denote one of S possible and mutually exclusive states of nature, $k = 1, 2, \ldots, S$, and p_k for the corresponding probabilities. We will use δ_j for $j = 0, 1$ to denote one of the two possible environmental policy settings between which a decision is being made. Individuals are assumed to be able to rank, *ex post*, realised outcomes according to a utility function of the form $U(Y, \delta_j \mid s_k)$ where Y denotes the individual's income as before, and where the | means 'given that' so that $U(Y, \delta_j \mid s_k)$ is the utility for some Y and δ_j given that some particular s_k obtains. The | symbol can also be read as 'conditional on', so that $U(Y, \delta_j \mid s_k)$ is the utility associated with the Y and δ_j conditional on the state s_k.

Let δ_0 represent one policy setting and δ_1 represent the alternative. Then

$$U(Y, \delta_1 \mid s_k) > U(Y, \delta_0 \mid s_k) \tag{13.7}$$

describes an *ex post* winner if δ_1 is adopted rather than δ_0, while

$$U(Y, \delta_1 \mid s_k) < U(Y, \delta_0 \mid s_k) \tag{13.8}$$

represents a loser. In either case *ex post* compensating surplus is defined by:

$$U(Y - CS_k, \delta_1 \mid s_k) = U(Y, \delta_0 \mid s_k) \tag{13.9}$$

Note that there is a k subscript on CS here – equation 13.9 defines compensating surplus given the kth state of nature. For a winner, CS_k is willingness to pay for the change of policy setting; for a loser, CS_k is willingness to accept compensation. Of course, the fact that an individual is a winner under a policy setting in one state of nature does not mean that he or she will be a winner under that policy in other states of nature. The expected value of compensating surplus, or expected compensating surplus, denoted E[CS] is the expectation of the compensated surpluses under each of the S possible states of nature:

$$E[CS] = \sum_{k=1}^{S} p_k CS_k \tag{13.10}$$

We can illustrate this in the Weisbrod park closure context. There are just two possible states of nature – s_1 where the individual wants to visit the park, and s_2 where she does not. The respective probabilities are p_1 and $1 - p_1$. Let δ_1 be the policy setting where the park is open (wilderness preserved) and δ_0 be the park closed (development allowed to go ahead). Then the individual is an *ex post* winner if the park is open and

$$U(Y, \delta_1 \mid s_1) > U(Y, \delta_0 \mid s_1)$$

[3] The discussion here largely follows that of Ready (1995).

with

$$U(Y - \mathrm{CS}_1, \delta_1 \mid s_1) = U(Y, \delta_0 \mid s_1)$$

defining CS_1, which would be WTP to have the park open. In this case CS_2 is zero because in the event that she does not want to visit the park, the individual will require no compensation for its closure. The individual is not a winner under δ_1, but neither is she a loser. Note that we are here assuming that the individual attaches no existence value to the park being open, to the wilderness remaining in an undeveloped state. In this case then,

$$\begin{aligned} \mathrm{E}[\mathrm{CS}] &= p_1 \cdot \mathrm{CS}_1 + (1 - p_1) \cdot \mathrm{CS}_2 \\ &= p_1 \cdot \mathrm{CS}_1 + (1 - p_1) \cdot 0 = p_1 \cdot \mathrm{CS}_1 \end{aligned}$$

Imagine the policy decision being taken repeatedly over time. Given that p_1 is the probability that the individual will suffer from a decision for closure, expected compensating surplus can be regarded as the average over many repetitions of her willingness to pay to avoid it.

Now consider matters *ex ante*, before the outcome is known, first in the general case. *Ex ante*, an individual's utility depends on the potential outcomes and their probabilities as assessed by that individual. If we use the ordinary utility function notation for *ex ante* utility,

$$U(Y, \delta_1) > U(Y, \delta_0) \tag{13.11}$$

simply means that before the outcome is known the individual prefers δ_1 to δ_0, so that if policy setting δ_1 were to eventuate she would, *ex post*, be a winner. On the other hand,

$$U(Y, \delta_1) < U(Y, \delta_0) \tag{13.12}$$

says that *ex ante* the individual prefers δ_0 to δ_1, so that if policy setting δ_1 were to eventuate she would, *ex post*, be a loser. Given this,

$$U(Y - \mathrm{OP}, \delta_1) = U(Y, \delta_0) \tag{13.13}$$

defines OP as the option price for δ_1. For the equation 13.11 case OP would be WTP, *ex ante*, for δ_1 rather than δ_0, while for the equation 13.12 case OP would be WTA compensation to accept δ_1 rather than δ_0.

Ex ante, the individual's utility is a function of the potential outcomes and their associated probabilities, that is,

$$U(Y, \delta) = \sum_{k=1}^{S} p_k U(Y, \delta \mid s_k) \tag{13.14}$$

This says that *ex ante* the utility associated with a Y/δ pair is the expected value of the *ex post* utilities that would go with that pair under different states of nature. Substituting from equation 13.14 into equation 13.13 we get

$$\sum_{k=1}^{S} p_k U(Y - \mathrm{OP}, \delta_1 \mid s_k) = \sum_{k=1}^{S} p_k U(Y, \delta_0 \mid s_k) \tag{13.15}$$

as defining OP for δ_1.

Consider the Weisbrod park policy decision again. We can define OP for the park staying open according to

$$U(Y - \mathrm{OP}, \delta_1) = U(Y, \delta_0) \tag{13.16}$$

or

$$p_1 U(Y - \mathrm{OP}, \delta_1 \mid s_1) = p_1 U(Y, \delta_0 \mid s_1) \tag{13.17}$$

Note that we do not find s_2 appearing in equation 13.17 for the reason discussed above. Note also that instead of the δ notation, we could use here the A/N notation that we used when first considering the question of option value using Figure 13.2. Equations 13.16 and 13.17 could, that is, be written as

$$U(Y - \mathrm{OP}, \mathrm{A}) = U(Y, \mathrm{N}) \tag{13.18}$$

and

$$p_1 U(Y - \mathrm{OP}, \mathrm{A} \mid s_1) = p_1 U(Y, \mathrm{N} \mid s_1) \tag{13.19}$$

To make this more concrete, let us consider a simple numerical example. Suppose that what determines whether the individual wants to visit the park or not on a weekend is the weather. In fine weather the individual will definitely want to go, while in bad weather she will definitely not want to go. Suppose that the park is open for free and that the individual's WTP for entry on a fine weekend is £10, and that the probability of fine weather is 0.5. Then, her E[CS] is £5. Now suppose that she is told that the park might be closed next weekend, then offered a ticket guaranteeing her access. On an actuarial basis, with no risk aversion, the value of the ticket is £5 = (0.5 × £10) + (0.5 × £0), E[CS]. If in order to avoid the risk of wanting to go to the park (fine weather) but not being able to (it is closed), the individual is WTP £6 for such a ticket, then OP = £6 and OV = £1.

According to this analysis, OV is not so much a separate category of preservation benefit as the difference between an *ex ante* measure, OP, and the expected value of an *ex post* measure, E[CS]. The question which arises is: which is the correct measure to use in ECBA? The consensus view emerging in the literature is that an *ex ante* measure is the right one. Essentially the basis for this is the acceptance of consumer sovereignty. In actually taking decisions concerning 'ordinary commodities', consumers proceed on an *ex ante* basis, and the argument is that the best measure of an individual's own preferences and attitude to risk for policy analysis is her own *ex ante* utility function that informs decisions about 'ordinary commodities'.[4]

Unfortunately, OP cannot be estimated from data on observable behaviour. However, E[CS] in some circumstances can be estimated from observable behaviour. In the Weisbrod park context, for example, and leaving aside the problems discussed in the previous chapter, one could use the TCM. If it were known that OV were positive, then from equation 13.6 it would follow that OP was greater than E[CS] and an estimate of E[CS] based on observable behaviour could be treated as a lower bound for OP. However, while Figure 13.2 suggests that risk aversion necessarily implies a positive OV, recent analysis shows that even for a risk-averse individual OV could in some circumstances be negative. Given this, E[CS] would not necessarily represent a lower bound for OP.

In principle, this need not be a major problem, as instead of trying to get at OP via observed behaviour, one could use the CVM with an appropriate *ex ante* scenario to directly elicit OP as WTP/WTA. In practice, the design of 'an appropriate *ex ante* scenario' – that is, one that effectively puts respondents in the intended hypothetical market and risk situation – is extremely difficult. We discussed in the previous chapter some of the problems with the CVM where respondents are put in situations where outcomes are to be treated as certain. These problems tend to be made worse when an effort to introduce risk into the scenario is undertaken, and there have been only a few CVM applications that have tried to elicit OP.

13.3 Risk and irreversibility

In the previous section we saw that, usually, for a risk-averse individual option price is greater than expected compensating surplus by an amount which is option value. To the extent that social decision making adopts the principle of consumer sovereignty, and given that most individuals are risk-averse, this leads to the conclusion that option price, rather than expected compensating surplus, should be used in ECBA. With respect to, for example, wilderness development, this suggests that the level that net development benefits have to attain to justify development is greater than would be the case in a world in which the future was certain. This conclusion is dependent on adopting a risk-averse position. In this section we consider arguments that work in the same direction, but do not require risk aversion. Since we are not assuming risk aversion, we can, and do, work with expected values, sums of money, rather than expected utilities.

The arguments which lead to an increase in the net development benefits required to justify development depend on the future being imperfectly known and on development being irreversible. Economic analysis usually assumes that allocation decisions are reversible. However, there is an implicit assumption in our previous discussions of wilderness development that once development occurs, it is irreversible. If this were not so, when the project ceased to do what it was intended for – when the mine was exhausted or the dam at the end of its safe life – then it would be possible, at some cost, to restore the wilderness, and this should be reflected in an ECBA. At least on a timescale relevant to human decision making, the assumption that once lost, the benefits of wilderness preservation are lost for ever, appears to be a reasonable approximation to the relevant stylised facts of wilderness development. A decision in favour of preservation, on the other hand, is clearly reversible.

[4] Ready (1995) argues for the use of *ex ante* OP from consideration of the properties of *ex ante* and *ex post* versions of compensation tests.

While the argument that we are interested in involves both irreversibility and risk, it will make things clearer to begin by considering just irreversibility alone.

13.3.1 Irreversibility with the future known

To introduce some of the implications of irreversibility, then, we begin with the, unrealistic, assumption that the future is known with certainty. We will consider a wilderness area yielding flows of amenity services, to be denoted *A*. We will assume that *A* is a function of the proportion of the wilderness area preserved from, say, logging. As the size of the area where logging is permitted increases, *A* falls. We consider benefits and costs as a function of *A*. The benefits are the preservation benefits, in terms of use and non-use values as discussed in the previous chapter, and we assume that marginal benefits decline as *A* increases. Consistently with our previous treatment, we assume that the costs of preservation as such are zero. However, we assume that preservation does entail forgone development benefits, which we treat here as costs of preservation. We assume that marginal costs increase with the area set aside from logging, and hence increase with *A*.

These assumptions are shown in Figure 13.3(a), where there is also shown as A^* the level of amenity service flow that goes with allocative efficiency. Figure 13.3(b) shows the corresponding behaviour for benefits minus costs, that is, net benefit, NB. Net benefits attain a maximum at A^*. Figure 13.3(c) shows the corresponding behaviour for the derivative of net benefit, which is marginal net benefit, MNB, which is zero at the level of A for which NB attains a maximum, A^*. $\text{MNB}(A) = 0$ is an alternative way of stating the necessary condition for maximum NB. In what follows here it will be convenient to work with MNB. The downward-sloping MNB function that we work with is a fairly generally appropriate assumption. The assumed linearity makes it possible, in Appendix 13.1, to do some simple algebra which supports the discussion in the text here.

We divide time into two periods, now and 'the future'. Now consider Figure 13.4, which, in (a) shows MNB_1 for period 1, now, and in (b) shows MNB_2 for period 2, 'the future'. Future net benefits are expressed in terms of their present value. As Figure 13.4 is drawn, even after discounting, period 2 MNB is greater than period 1 MNB for a given level of the service flow. Generally, this reflects the considerations advanced in Chapter 11 in connection with the Krutilla–Fisher model concerning the

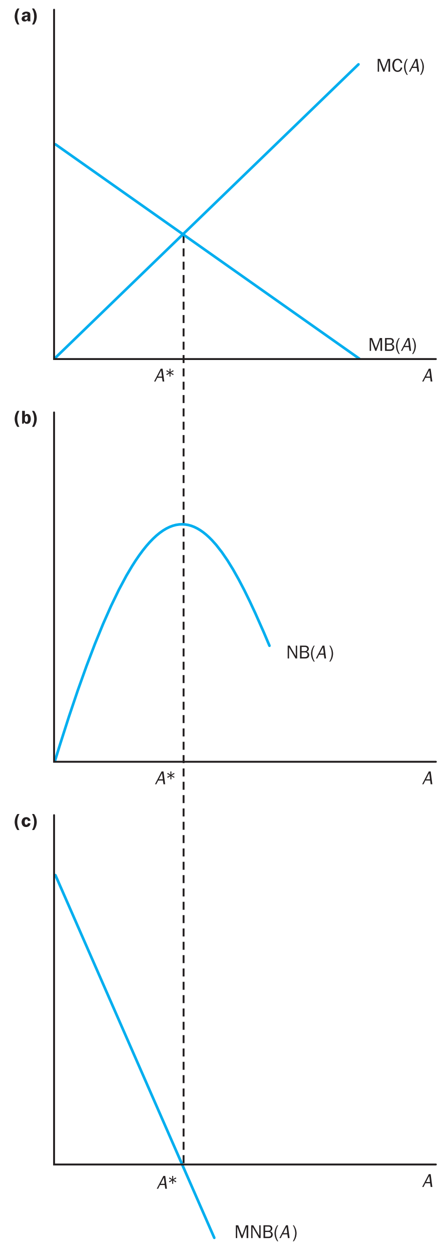


Figure 13.3 Alternative ways of identifying maximum net benefit

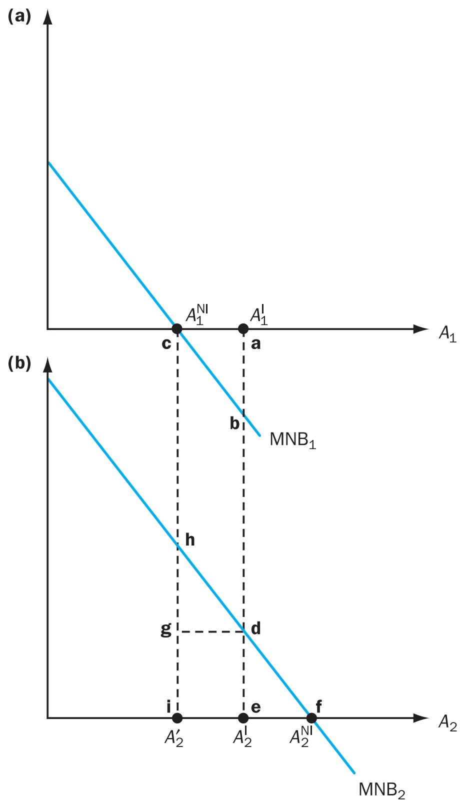


Figure 13.4 Irreversibility and development with the future known

relative prices of environmental amenity services and produced commodities. Particularly, in Figure 13.4 MNB_2 has the same slope as MNB_1 but a larger intercept.

Consider now the level of amenity service flow in each period that goes with current and intertemporal allocative efficiency if there is no irreversibility. Given that we are dealing with period 2 in terms of appropriately discounted net benefit, in the absence of irreversibility, an efficient outcome would involve choosing a consumption level for environmental amenity services in each period for which MNB_1 and MNB_2 are equal to zero, A_1^{NI} and A_2^{NI}. Note that $A_2^{NI} > A_1^{NI}$.

Now assume that development is irreversible. How does this affect things? It constrains the choices in the two periods such that A_2 cannot be greater than A_1. If at the outset a level of period 1 development A_1^{NI} were chosen, myopically ignoring irreversibility, then period 2 A could be at most A_2'. If the decision on the period 1 level of development were taken in the light of the irreversibility constraint, the outcome would be A_1^I and A_2^I. As compared with the myopic decision-making outcome, taking account of irreversibility means a higher level of A (less development) in period 1 and period 2.

Taking irreversibility into account means, as compared with the situation where irreversibility is ignored, incurring costs in period 1 so as to secure benefits in period 2. The period 1 costs arise from selecting a level of A_1 above that where MNB_1 is equal to 0. The period 2 benefits resulting are due to a level of A_2 for which MNB_2 is nearer to 0 than at A_2'. For efficiency, costs and benefits must be equal at the margin. This is the case in Figure 13.4, where **ab**, MNB_1 at A_1^I, is equal to **de**, MNB_2 at A_2^I. Taking the irreversibility constraint into account leads to an outcome where MNB_1 and MNB_2 are equal but of opposite sign. Recall that MNB_2 refers to period 2 net benefits considered in present-value terms in period 1.

As compared with a situation where development is reversible, irreversibility entails costs. In Figure 13.4 these costs are given by comparing A_1^{NI} with A_1^I and A_2^{NI} with A_2^I. In period 1 the cost is given by the area of the triangle **abc**, and in period 2 by the area of triangle **def**. If there is irreversibility, ignoring it entails costs. Given irreversibility, the efficient outcome is A_1^I/A_2^I, but if irreversibility is ignored the actual outcome will be A_1^{NI}/A_2'. Ignoring irreversibility leads to a gain in period 1 given by the area of triangle **abc**, but to a loss in period 2 given by the area **edhi**. The loss is greater than the gain, so that there is a net cost to ignoring irreversibility.

We have been assuming here that the future is known with certainty. In reality, when considering such matters as wilderness development, irreversibility is combined with imperfect future knowledge. We next use this simple framework for considering the implications of irreversibility taking account of imperfect knowledge of the future. In order to do that we use the analysis of decision making given imperfect knowledge of the future introduced in the first section of this chapter.

13.3.2 Irreversibility in a risky world

Figure 13.5(a) is the same as Figure 13.4(a), apart from the appearance of A_1^{IR}, to be explained. Figure 13.5(b)

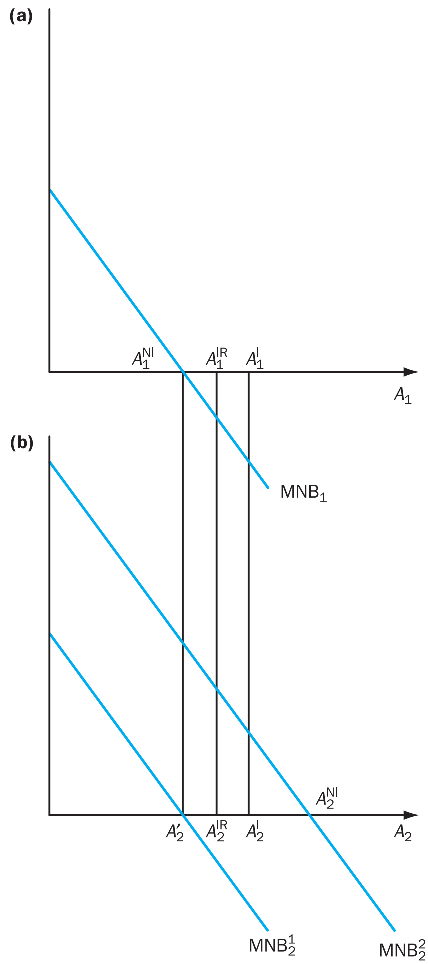


Figure 13.5 Irreversibility and development with imperfect future knowledge

shows the same MNB function as Figure 13.4(b), but here it is labelled MNB_2^2 instead of just MNB_2. The superscript 2 now appears because we also have MNB_2^1, which has the same intercept and slope as MNB_1 in Figure 13.5(a). We are now considering a situation where MNB as a function of A is known for period 1, but where the decision maker does not know for period 2 which of two MNB functions will eventuate, MNB_2^1 which is the same as MNB_1, or MNB_2^2, which has the same slope as MNB_1 but a larger intercept. While it is not known, when deciding on the level of period 1 development and hence the level of A_1, which of MNB_2^1 or MNB_2^2 will obtain in period 2, the decision maker can assign probabilities p to MNB_2^1 and $q = (1 - p)$ to MNB_2^2.

In Figure 13.5 A_1^{NI} is the same level of A_1 as A_1^{NI} in Figure 13.4 and both refer to the outcome of decision making which ignores irreversibility. Given irreversibility, A_2 must equal (strictly be no greater than) A_1, so A_1^{NI} the same in both figures implies A'_2 the same in both. A_1^{I} and A_2^{I} in Figure 13.5 are also the same as in Figure 13.4, and refer to the outcome where there is irreversibility but no risk and it is known that the period 2 MNB function will be MNB_2^2 which is the same as MNB_2 in Figure 13.4. A_1^{IR} and A_2^{IR} are the outcomes for a decision-making process that takes on board both irreversibility and risk, and adopts risk neutrality. In this case, adding imperfect future knowledge about MNB to irreversibility leads to lower levels of amenity – higher levels of development – than irreversibility alone, but which are higher – lower levels of development – than would have resulted if irreversibility were ignored.

This is established in Appendix 13.2. The results are reasonably intuitive. If irreversibility is ignored, then in the first period the level of A_1 can be chosen by setting MNB_1 equal to zero, and the fact that which of MNB_2^1 or MNB_2^2 will eventuate is unknown is irrelevant. Given that irreversibility is a fact, though ignored, the choice of the period 1 level of A immediately gives its period 2 level, and we get A_1^{NI} and A'_2. If the decision-making process recognises irreversibility and assumes MNB_2^2, the situation is as discussed for Figure 13.4 and we get A_1^{I} and A_2^{I}. Where it also recognises that period 2 might involve MNB_2^2 or MNB_2^1, it uses the weighted average of these two alternatives, with weights that are the assigned probabilities, and ends up in an intermediate situation, A_1^{IR} and A_2^{IR}.

13.3.3 Quasi-option value

We now consider the implications of irreversibility in a world where there is imperfect knowledge of the future, but where more knowledge will become available after a decision has been made. We again look at the matter of wilderness development. This is the context in which Arrow and Fisher (1974) introduced the concept of quasi-option value, and our treatment follows that of Arrow and Fisher quite closely. To simplify, we now consider a situation where development is ‘all or nothing’ in the sense that either development occurs and drives wilderness amenity benefits to zero, or development does not occur. This is like the decision considered by Weisbrod – either the national park is permanently closed (to allow development), or it remains open.

The essential point that this special formulation makes clear is that where there is the prospect of improved information 'the expected benefits of an irreversible decision should be adjusted to reflect the loss of options it entails' (Arrow and Fisher, 1974, p. 319). The adjustment is required even if the decision maker is risk-neutral. The size of the adjustment is quasi-option value. While we discuss quasi-option value in an all-or-nothing development context, the basic idea carries over to situations where the wilderness area can be partially developed – indeed, it carries over to any situation where one course of action is irreversible and where there will in the future be improved information about the future situation.

As before, time is divided into two periods, 1 being 'now' and 2 'the future'. The decision maker has complete knowledge of all relevant period 1 conditions. At the start of period 1, period 2 outcomes can be listed and probabilities attached to them. A decision involving irreversible consequences must be taken at the start of period 1. At the end of period 1, complete knowledge about period 2 will become available to the decision maker.

The decision to be taken at the start of period 1 is whether to permit development of a wilderness area. The options are shown in Table 13.1. As before, D is for development, P is for preservation, and period 2 costs and benefits are to be understood as discounted present values. R^i is the return associated with the ith option, B_{pt} is preservation benefits, B_{dt} is development benefits, C_{dt} is development costs, which are treated as arising only in the period in which the development project is undertaken, and as before we do not explicitly distinguish preservation costs. Option 1 involves initiating development at the start of period 1, and given irreversibility development in period 1 implies development in period 2. Hence, option 4, having the area developed in 1 but preserved in 2, is shown in Table 13.1 as infeasible. The operative alternatives to having the area in a developed state in both periods are option 2 – preservation followed by development at the start of period 2 – and option 3 – never develop.

Table 13.1 Two-period development/preservation options

Option	Period 1	Period 2	Return
1	D	D	$R^1 = (B_{d1} - C_{d1}) + B_{d2}$
2	P	D	$R^2 = B_{p1} + (B_{d2} - C_{d2})$
3	P	P	$R^3 = B_{p1} + B_{p2}$
4	D	P	Is infeasible

Let us label the return to the decision taken at the start of period 1 to proceed immediately with development R^d, so that:

$$R^d = R^1 = (B_{d1} - C_{d1}) + B_{d2} \tag{13.20}$$

The return to the decision taken at the start of period 1 to preserve is either R^2 or R^3, depending on whether or not development is initiated at the start of period 2 given the information then available. If B_{p2} then is known to be bigger than $B_{d2} - C_{d2}$, the area will be preserved in period 2, giving R^3. If $B_{d2} - C_{d2}$ is then known to be bigger than B_{p2}, development will be undertaken at the start of period 2, giving R^2. We can express this as

$$R^p = B_{p1} + \max\{B_{p2}, (B_{d2} - C_{d2})\} \tag{13.21}$$

where R^p is the return to the period 1 decision for preservation, and the right-hand side is to be read as B_{p1} plus whichever is the greater of B_{p2} and $(B_{d2} - C_{d2})$ – 'max' is short for 'the largest of the terms appearing inside the braces'. Note that B_{p1} is common to both R^2 and R^3.

Now, suppose for the moment that the decision maker does have complete knowledge of the relevant future circumstances, that at the start of period 1 he or she knows all the B_{pt}, B_{dt} and C_{dt}. Then the decision maker also knows R^d and R^p, and the decision will be to go ahead with development immediately if $R^d > R^p$, which is if $R^d - R^p > 0$, which on substituting from equations 13.20 and 13.21 is

$$(B_{d1} - C_{d1}) + B_{d2} - B_{p1} - \max\{B_{p2}, (B_{d2} - C_{d2})\} > 0 \tag{13.22}$$

which can be written

$$N_1 + B_{d2} - \max\{B_{p2}, (B_{d2} - C_{d2})\} > 0 \tag{13.23}$$

where $N_1 = (B_{d1} - C_{d1}) - B_{p1}$. In other words, N_1 is that which would actually be known to the decision maker at the start of period 1.

The other terms in the expression 13.23 could not, in fact, be known to the decision maker at the start of period 1, so 13.23 is not an operational decision rule. We are, however, assuming that the possible outcomes for B_{d2}, B_{p2} and $(B_{d2} - C_{d2})$ are known to the decision maker and that he or she can attach probabilities to the mutually exclusive outcomes. In that case, it is tempting to simply replace known outcomes in the expression 13.23 by the corresponding expectations, or expected values, and to write an operational decision rule as: go ahead with development at the start of period 1 if

$$\mathrm{N}_1 + \mathrm{E}[B_{d2}] - \max\{\mathrm{E}[B_{p2}], \mathrm{E}[(B_{d2} - C_{d2})]\} > 0 \tag{13.24}$$

However, using this rule ignores the fact that more information will be available at the start of period 2. If the area is developed at the start of period 1 this information cannot be used, since the area will necessarily be in a developed state in period 2. If the area is not developed at the start of period 1, the new information could be used at the start of period 2 to decide between development and preservation then.

The proper decision rule is one that takes this on board, as the expression 13.24 does not. Now, of course, a decision has to be taken at the start of period 1, and the decision maker does not then have the information that will become available at the start of period 2. But, by assumption, the decision maker does at the start of period 1 know what the informational possibilities are and the probabilities to attach to outcomes in that respect. So, he or she could use the decision rule: go ahead with development at the start of period 1 if

$$N_1 + \mathrm{E}[B_{d2}] - \mathrm{E}[\max\{B_{p2}, (B_{d2} - C_{d2})]\} > 0 \tag{13.25}$$

Whereas in expression 13.24 the decision maker uses the maximum of the expected values of period 2 preservation benefits and net development benefits, in 13.25 he or she uses the expectation of the maximum of period 2 preservation benefits and net development benefits. The left-hand side of expression 13.25 will be larger than the left-hand side of 13.24, so that the former decision rule is a harder test for development to pass at the start of period 1. The difference between the left-hand sides of expressions 13.25 and 13.24 is quasi-option value. It is the amount by which a net development benefit assessment which simply replaces outcomes by their expectations should be reduced, given irreversibility, to reflect the pay-off to keeping options open, by not developing, until more information about future conditions is available.

This analysis can be illustrated with a simple numerical example. Suppose that there are just two possible period 2 situations, A and B, differentiated only by what the preservation benefits for 'the future' will be learned to be. B_{d2} and C_{d2} are the same for A and B, and for both $(B_{d2} - C_{d2}) = 6$. For A, B_{p2} is 10; for B, B_{p2} is 5. At the beginning of period 1, A and B are seen as equiprobable so that $p^A = p^B = 0.5$. In this case, for 13.24 we have for the third term to the left of the > sign

$$\begin{aligned}&\max\{\mathrm{E}[B_{p2}], \mathrm{E}[(B_{d2} - C_{d2})]\} = \max\{[(0.5 \times 10)\\ &+ (0.5 \times 5)], [(0.5 \times 6) + (0.5 \times 6)]\}\\ &= \max\{7.5,6\} = 7.5\end{aligned}$$

so the development will get the go-ahead if

$$N_1 + \mathrm{E}[B_{d2}] - 7.5 > 0 \tag{13.26}$$

Now consider 13.25. We have two possible outcomes:

A where $B_{p2} > (B_{d2} - C_{d2})$, $B_{p2} = 10$, $p^A = 0.5$

B where $B_{p2} < (B_{d2} - C_{d2})$, $(B_{d2} - C_{d2}) = 6$, $p^B = 0.5$

Hence,

$$\begin{aligned}\mathrm{E}[\max\{B_{p2}, (B_{d2} - C_{d2})\}] &= (0.5 \times 10) + (0.5 \times 6)\\ &= 8\end{aligned}$$

and following this decision rule, development will get the go-ahead if

$$N_1 + \mathrm{E}[B_{d2}] - 8 > 0 \tag{13.27}$$

Suppose $N_1 + \mathrm{E}[B_{d2}] = 7.75$. Then, using 13.24/13.26 development would be decided on at the start of period 1, while using 13.25/13.27 the decision would be to preserve in period 1. The test based on 13.25 is harder to pass than the 13.24-based test. As compared with 13.24, 13.25 adds a premium to the value for $N_1 + \mathrm{E}[B_{d2}]$ required to justify a decision for development at the start of period 1. That premium is quasi-option value, which in this example is $0.5 = 8 - 7.5$.

Positive quasi-option value is a general result. This is straightforward, and instructive, to establish where $E[B_{p2}] > E[(B_{d2} - C_{d2})]$, as in this numerical example.[5] Consider first 13.24 under that assumption, in which case it becomes

$$N_1 + E[B_{d2}] - E[B_{p2}] > 0 \tag{13.28}$$

Now consider $\max\{B_{p2}, (B_{d2} - C_{d2})\}$ from 13.25. This is either B_{p2} or a number larger than B_{p2}. So long as the possibility that $(B_{d2} - C_{d2}) > B_{p2}$ is entertained by the decision maker, $E[\max\{B_{p2}, (B_{d2} - C_{d2})\}]$ will be greater than $E[B_{p2}]$, and 13.25 which is

$$N_1 + E[B_{d2}] - E[\max\{B_{p2}, (B_{d2} - C_{d2})\}] > 0$$

will be a harder test to pass than 13.28, with

$$E[\max\{B_{p2}, (B_{d2} - C_{d2})\}] - E[B_{p2}] = QOV$$

where QOV is for quasi-option value.

The basic point about the existence of quasi-option value is that, where more knowledge about future conditions will become available after an irreversible decision has been made, even with risk neutrality, simply replacing random variables with their expectations and then optimising will lead to the wrong decision. Recall that in Figure 13.5 (and see also Appendix 13.2) we found that adding imperfect future knowledge to irreversibility led to a higher current level of development. We did not there incorporate any quasi-option value into the, risk-neutral, decision-making procedure, because we did not assume that more information would become available at the end of period 1. In general the assumption of increased information in the future would be more appropriate than the contrary assumption, and social decision making should include quasi-option value. However, it is clear from the discussion here that in order to estimate quasi-option value it would be necessary to know, or to assume, a lot about possible outcomes, their current probabilities and the prospects for additional information in the future. In practice ECBA rarely takes any account of quasi-option value in any formal quantitative way.

13.4 Environmental cost–benefit analysis revisited

Let us briefly bring together some of the foregoing ideas in the following context. An area of completely undeveloped wilderness land is currently privately owned. A large mineral deposit has been discovered and the owner plans to open a mine. The government is considering purchasing the land and making it a national park so as to preserve the amenity and life-support services that the area provides in its undeveloped state. If the mine went ahead there would be start-up costs with a present value of £20 million, to open the mine and construct the necessary infrastructure. Once operational the mine would yield net revenues of £6 million for 100 years, after which the ore body would be exhausted. Using the standard result for the present value of £*x* per annum for ever as an approximation, at an interest rate of 5%, £6 million for 100 years has a present value of £120 million. Hence, we have NPV′ for the mine equal to £100 million. This is the capitalised value of the area used for mining, the opportunity cost of the creation of a national park.

Consider first inverse ECBA. What is the minimum value of the environmental services yielded by the land in its undeveloped state that would justify stopping the mine and creating the park? We will assume that the only cost involved in the park option is the cost of acquiring the land. In that case, the answer is £100 million, in present-value terms. If the relevant population were collectively willing to pay £100 million now, or more, the government would on standard allocative efficiency grounds be justified in buying the land to create a national park. The per capita test for going ahead with the park would be: is the average member of the population willing to pay a once-off sum now of £(100/*N*), where *N* is the population size in millions? For a population of 20 million, roughly the number of adults in the UK, this is a lump sum of £5. This can be converted to an equivalent annual payment for a given number of years.

[5] The general result is established in Arrow and Fisher (1974). See also Fisher and Hanemann (1986).

Actually, if the arguments of Krutilla and Fisher about the relative prices of wilderness services and 'ordinary' commodities, considered in Chapter 11, are accepted, this way of calculating the threshold value for the value of environmental services lost if the mine goes ahead is biased in favour of the mine. Equation 11.29 can be read as saying that for the mine

$$\text{NPV} = 0 \text{ if } \text{NPV}' = P/(r - a)$$

where for the present example NPV′ is £100 million and $r = 0.05$. Suppose that P, the initial value for preservation benefits, is taken to be just £1 million per annum. What would a, the growth rate for the value of environmental services relative to ordinary commodities, then have to be to justify creating the park? Solving

$$100 = 1/(0.05 - a)$$

gives $a = 0.04$.

So, if it were known that the mine's NPV was £100 million, that current population willingness to pay for the services of the area undeveloped was £1 million, that development would be irreversible in its impact on those services, and that their relative value would grow by 4% per annum or more, then the government would be justified in acquiring the land and creating the park. Of course, in fact, things are not known in this way, and the question that arises is: how should ECBA take account of risk? It is tempting to answer that it is simply a matter of replacing known outcomes by their expected values. In that case, using continuous time notation for convenience, instead of calculating NPV according to equation 11.26,

$$\text{NPV} = \int_0^T \{B(\text{D})_t - C(\text{D})\}\text{e}^{-rt}\text{d}t - \int_0^T \{B(\text{P})_t\}\text{e}^{-rt}\text{d}t$$

we would use the probabilities for the various possibilities in regard to B(D), C(D) and B(P) to calculate

$$\text{E[NPV]} = \int_0^T \{\text{E}[B(\text{D})]_t - \text{E}[C(\text{D})]_t\}\text{e}^{-rt}\text{d}t - \int_0^T \{\text{E}[B(\text{P})]_t\}\text{e}^{-rt}\text{d}t \qquad (13.29)$$

However, if individuals are risk-averse and this is to be reflected in ECBA, this is incorrect. Some allowance for risk aversion must be made. Then the proper test for the mining project, and for the creation of the park, is

$$\text{NPV}^* = \int_0^T \{\text{E[B(D)]}_t - \text{E}[C(\text{D})]_t\}\text{e}^{-rt}\text{d}t - \int_0^T \{\text{E}[B(\text{P})]_t\}\text{e}^{-rt}\text{d}t - \int_0^T \text{CORB}_t\text{e}^{-rt}\text{d}t \qquad (13.30)$$

where CORB_t is the cost of risk bearing at time t. It is sometimes suggested that risk can be dealt with by using expected values and an increased rate of discount, as in

$$\text{NPV}^{**} = \int_0^T \{\text{E}[B(\text{D})]_t - \text{E}[C(\text{D})]_t\}\text{e}^{-(r+b)t}\text{d}t - \int_0^T \{\text{E}[B(\text{P})]_t\}\text{e}^{-(r+b)t}\text{d}t \qquad (13.31)$$

where b is the 'risk premium' to be added to the standard discount rate. Apart from the problem of deciding on a proper value for b, this implies that the cost of risk bearing is decreasing exponentially over time. This can be seen by writing equation 13.31 as

$$\text{NPV}^{**} = \int_0^T \{\text{E}[B(\text{D})]_t - \text{E}[C(\text{D})]_t\}\{\text{CORBe}^{-bt}\}\text{e}^{-rt}\text{d}t - \int_0^T \{\text{E}[B(P)]_t\}\{\text{CORBe}^{-bt}\}\text{e}^{-rt}\text{d}t$$

where CORB is some initial value for the cost of risk bearing, which thereafter declines at the rate b. While the assumption of an exponentially decreasing cost of risk may be appropriate in some cases, it clearly is not generally valid. The correct way to proceed is to estimate CORB_t over the life of the project and incorporate those estimates as in equation 13.30.

It has been argued (see Arrow and Lind, 1970) that when a project is undertaken by government on behalf of society as a whole, no allowances for risk need be made. The reasoning here is that when government undertakes risky projects, the risks are spread (or pooled or diversified) over many individuals. In aggregate, therefore, there is no risk attached to collectively undertaken investments, and no need to include estimates of CORB_t. This argument is not now accepted as being applicable in the context of projects that have environmental impacts. The reason for this is that environmental services are typically public goods – they are non-rival and non-excludable, so that the assumption of risk spreading does not hold.

The problem then remains of estimating CORB_t. As we have seen, this is now looked at under the 'option-value' rubric – in the ECBA context option-value is the cost of risk bearing. What this means is that, leaving aside existence values for the moment, an ECBA which does not simply ignore risk should use either 13.30 with $\text{E}[B(\text{P})]_t$ replaced by $\text{E}[\text{CS}]_t$ and CORB_t replaced by OV_t, or, more directly, replace both $\text{E}[B(\text{P})]_t$ and CORB_t with OP_t, where CS is compensating surplus and OP is option price. It should use, that is, either

$$\text{NPV}^* = \int_0^T \{\text{E}[B(\text{D})]_t - \text{E}[C(\text{D})]_t\}\text{e}^{-rt}\text{d}t - \int_0^T \{\text{E}[\text{CS}]_t\}\text{e}^{-rt}\text{d}t - \int_0^T \text{OV}_t\text{e}^{-rt}\text{d}t \quad (13.32)$$

or

$$\text{NPV}^* = \int_0^T \{\text{E}[B(\text{D})]_t - \text{E}[C(\text{D})]_t\}\text{e}^{-rt}\text{d}t - \int_0^T \text{OP}_t\text{e}^{-rt}\text{d}t \quad (13.33)$$

In principle, as previously noted, OP could be obtained directly from suitable application of the CVM. In practice this is difficult. It is understood that in most cases OV will be positive, so that if equation 13.32 is used with OV set at zero, NPV* will be overestimated. Of course, as discussed in the previous chapter, what actually gets estimated by, for example, the TCM, is Marshallian consumer surplus, MCS, rather than the theoretically correct Hicksian measure. The relationship between MCS and CS where it is quantity/quality change, rather than price change, that is at issue is, as discussed in the previous chapter, complex and they may diverge widely in unknown ways.

What all this means is that while at a theoretical level the proper way to do ECBA accounting for risk aversion is clear enough, the practical implementation of the procedures is difficult. In practice, an ECBA exercise of the kind we are considering here would use

$$\text{NPV} = \int_0^T \{B(\text{D})_t - C(\text{D})_t\}\text{e}^{-rt}\text{d}t - \int_0^T \{B(\text{P})_t\}\text{e}^{-rt}\text{d}t$$

where $B(\text{P})$ would include estimated use and existence value for the relevant population, and then subject the central-case result to sensitivity analysis. ECBA practitioners are aware of the importance of option value and quasi-option value, but find it hard to put numbers to these concepts. It is rather a matter of considering whether plausible variations to the central-case numbers can produce a negative NPV, when some judgemental allowance is made for option and quasi-option value.

13.5 Decision theory: choices under uncertainty

In the first section of this chapter we made the distinction between risk and uncertainty. Thus far we have been considering individual and social decision making in circumstances of risk, that is where the decision maker proceeds on the assumption that he or she can enumerate all possible future states and assign probabilities to them. We now consider decision making in the face of uncertainty, where the decision maker enumerates all possible states relevant to the decision but cannot attach probabilities to those states. The approach that we adopt is called 'decision theory' and is a branch of the theory of games.

We focus on social decision making and treat what we shall call 'society' as one of the players of a game. The other 'player' is by convention called 'nature' and the games that we shall be considering are often called 'games against nature'. Society must select a move (or strategy) in ignorance about which state of nature will occur, and is unable to attach probabilities to states of nature. However, it is assumed that society can estimate its 'pay-off matrix'. The pay-off matrix is a statement of the alternative strategies open to society, the possible states of nature, and the pay-offs associated with each combination of strategy and state of nature. Table 13.2 is an example of a pay-off matrix constructed for the context that we considered in the previous section of the chapter – the decision as to whether to stop

Table 13.2 A pay-off matrix

	C	D	E
A Conserve the wilderness area as a national park	120	50	10
B Allow the mine to be developed	5	30	140

development in a wilderness area by making it a national park.

There are two strategies, A and B, and three possible states of nature C, D and E. The entries in Table 13.2 are millions of £s of NPV associated with the corresponding strategy–state-of-nature combination. If state C eventuates and the park exists, a large number of individuals choose to visit the park and enjoy its wilderness amenities, while if the mine is allowed to go ahead it turns out that it is a commercial failure. State E is the converse of this: if the park exists few individuals choose to visit it and the value attached to its existence by non-visitors is low, while the mine turns out to be very successful commercially. The state of nature D is intermediate between C and E: the mine is moderately successful if it goes ahead, as is the park if it does.

If C eventuates, society is richly rewarded *ex post* for adopting strategy A, receiving a pay-off of 120. In contrast, its returns to strategy B are very poor, obtaining a pay-off of just 5. The remaining four cells in the matrix show society's pay-offs from A or B when the state of nature is either D or E. Note that the best of all possible outcomes is that given by the lower right cell, where society allows the mine to go ahead and it is commercially successful.

Which strategy, A or B, should the government acting on behalf of society select? Should it allow the mine to proceed or create a national park? Let us examine four of the decision rules that have been proposed for games against nature.

1) Maximin rule

Government sets out the pay-off matrix and selects the strategy with the least-bad worst outcome. The label 'maximin' signifies that the strategy that maximises the minimum possible outcome is selected. Inspection of the pay-off matrix in Table 13.2 reveals A to be the maximin strategy. If B is selected, the worst possible outcome is 5, while if A is selected the worst outcome is 10.

While avoiding worst possible outcomes has some attraction, the maximin rule can easily lead to choices which seem to contradict common sense. This arises because the maximin rule ignores most of the information in the pay-off matrix. In particular, the pay-offs in best possible cases are ignored. Moreover, the maximin decision rule means that decisions are made entirely on the basis of the most adverse possibilities. If one strategy was only marginally better than a second in terms of its worst outcome, the first would be preferred no matter how much more preferable the second may be under all other states of nature.

2) Maximax rule

In contrast to what may be regarded as the very cautious maximin strategy, the 'maximax' decision rule is very adventurous. Each available strategy is examined to identify its best outcome. The one selected is that with the best of the best outcomes. This rule implies that government should adopt strategy B, as its best outcome is 140, in state E, whereas the best outcome from adopting A is 120, in state C.

The maximax rule suffers from a similar weakness to the maximin rule, as it ignores most of the information in the pay-off matrix. In this case, all pay-offs other than the best possible ones are ignored. Once again, the choices implied by this rule can fly in the face of common sense.

3) Minimax regret rule

The essence of the 'minimax regret' rule is the avoidance of costly mistakes. To implement this rule, a regret matrix is derived from the pay-off matrix. This is done by identifying for each state of nature the strategy with the largest pay-off and then expressing all the other pay-offs for that state of nature as deviations from the largest. The entries in the regret matrix are the difference between the actual pay-off for the strategy–state-of-nature combination and what the pay-off would have been if the best strategy for the state of nature had been chosen. The regret matrix for our illustrative example is shown in Table 13.3.

Table 13.3 A regret matrix

	C	D	E
A	0	0	130
B	115	20	0

Once the regret matrix has been calculated, government plays a minimax game using these regrets. Each row of the regret matrix is examined to identify the largest possible regret. The strategy with the lowest of the largest regrets is then chosen. The minimax regret rule leads to selection of B in this example, as its most costly mistake is 115 in state C, whereas for A the most costly mistake is 130 in state E.

4) Assignment of subjective probabilities

According to 'the principle of insufficient reason', in the absence of any better information the decision maker should assign equal probabilities to the mutually exclusive outcomes, and adopt the strategy that then has the pay-off with the greatest expected value. For our example, this leads to selection of strategy A, which has an expected value of 60 whereas the expected value for B is 58.33.

The equal probabilities are just subjective probabilities. In some situations there may be information available which enables the assignation of unequal subjective probabilities. Unlike the previous decision rules, selecting the strategy with the largest expected value on the basis of subjective probabilities does consider all alternatives in the pay-off matrix.

It is clear from this brief exposition of decision theory that while it can provide insights into how decision makers might behave in the face of uncertainty, it cannot tell us which is the best way to make choices in an uncertain world. Indeed, the concept of rational behaviour is problematic in the face of uncertainty – there is no way of making decisions that can be unambiguously identified as doing the best for the decision maker in the relevant circumstances. It may be that this is why many economists are uncomfortable with uncertainty, and prefer to deal with situations where the assignment of 'objective' probabilities is impossible by treating decision makers as assigning 'subjective' probabilities.

13.6 A safe minimum standard of conservation

In the first section of this chapter we defined radical uncertainty as a situation in which the decision maker is not able to list all of the possible outcomes. This is the context in which the idea of the 'safe minimum standard' was originally formulated. To see what is involved, let us modify the illustrative example just considered. Let us confine the possibilities to state E, but suppose that it is known that the construction of the proposed mine will mean that a population of some plant will be destroyed, and that this population is thought to be the only one in existence. The mine proponents have undertaken to attempt the re-establishment of the plant in another location, and the cost of so doing is included in their project appraisal. It is unknown whether the attempt will be successful.

To consider the park/mine decision now, we can specify just two states of nature, F and U: F stands for 'favourable' and is where the relocation is successful, while U is for 'unfavourable', in which state of nature the relocation is unsuccessful. What payoff should be assigned to having the mine go ahead if the state U eventuates? The problem is that not only is it impossible to assign probabilities to success and failure – it has never been tried before – but that it is also unknown whether any other populations of the plant species exist. In this case the state of nature U may imply extinction of the species. In this case, the regret matrix is then as shown in Table 13.4, where z is an unknown number. For state F the regrets in Table 13.4 are the same as in Table 13.3. For state U, the park option A would avoid the species extinction, whereas B could entail that extinction, which carries a large, but unknown, regret, z.

In fact, as Table 13.4 is constructed, it is implicit, by virtue of the 0 entry for A/U, that although we have said that z is an unknown number, it is supposed

Table 13.4 A regret matrix for the possibility of species extinction

	F	U
A	130	0
B	0	z

large enough that A, stop the mine, is the correct strategy in state U. But this presumption does not, following minimax regret, indicate which strategy to adopt. One way to proceed would be to assume that although the cost of species extinction is unknown, it can be presumed large enough to make A the right strategy. It is exactly this presumption that underlies the idea of the safe minimum standard as originally put forward. As made by Bishop (1978) the argument for this presumption is as follows. Species extinction involves an irreversible reduction in the stock of potentially useful resources which is the existing portfolio of species. In a state of radical uncertainty there is no way of knowing how large the value to humans of any of the existing species might turn out to be in the future. Two kinds of ignorance are involved here. First, there is social ignorance about future preferences, needs and technologies. Second, there is scientific ignorance about the characteristics of existing species as they relate to future social possibilities and needs. The extinction of any species is, therefore, to be presumed to involve future costs that may be very large, even when discounted into present-value terms. The argument here is essentially that the species that may become extinct may turn out to be one for which there is no substitute.

Applying the safe minimum standard for conservation (SMS) criterion as stated to projects which could entail species extinction would mean rejecting all such projects. All that is required for rejection is the possibility that going ahead with the project could involve species extinction. SMS is a very conservative rule. It means forgoing current gains, however large, in order to avoid future losses of unknown, but presumed very large, size. A modified SMS has been proposed according to which the strategy that ensures the survival of the species should be adopted, unless it entails unacceptably large costs. This is less conservative, but leaves it to be determined whether any given cost is 'unacceptably large'.

The SMS criterion can also be applied to target setting for pollution policy, where it would imply that targets should be set so that threats to the survival of valuable resource systems from pollution flows are eliminated, provided that this does not entail excessive cost. Alternatively, one may view the SMS criterion here in terms of constraints – pollution policy should in general be determined using an efficiency criterion, as discussed in Chapter 5, but subject to the overriding constraint that an SMS is satisfied. This formulation of pollution policy recognises the importance of economic efficiency but accords it a lower priority than conservation when the two conflict, provided that the opportunity costs of conservation are not excessive. This compromise between efficiency and conservation criteria implies that 'correct' levels of pollution cannot be worked out analytically. Instead, judgements will need to be made about what constitutes excessive costs, and which resources are deemed sufficiently valuable for application of the SMS criterion.

We have discussed the problem of climate change due to increasing atmospheric concentrations of greenhouse gases owing to human activity at several points earlier in the book: see Chapter 9 especially. An essential feature of this problem is uncertainty. In Box 13.1 we look at how this was dealt with in an important recent review of the climate change problem conducted for the UK government.

Box 13.1 How the Stern Review deals with risk and uncertainty as it attends the climate change problem

Figure 13.6 is a simple schematic representation of the structure of the enhanced greenhouse effect, which is the basis for the climate change problem. The dashed lines represent feedback effects. Biospheric changes driven by climate change affect the way climate responds to greenhouse gas concentrations, and the operation of sinks. Changes in human behaviour resulting from climate change affect economic activity. The important point is that our knowledge of all the links depicted in Figure 13.6 is highly imperfect. This is a key feature of the climate change problem, in terms both of objectives and instruments for responding to it. In this box we are interested in the former, and particularly the analysis of the mitigation objective.

The 1992 United Nations Framework Convention on Climate Change, the legal basis for international action on the problem, adopted a Safe Minimum Standard approach

Box 13.1 *continued*

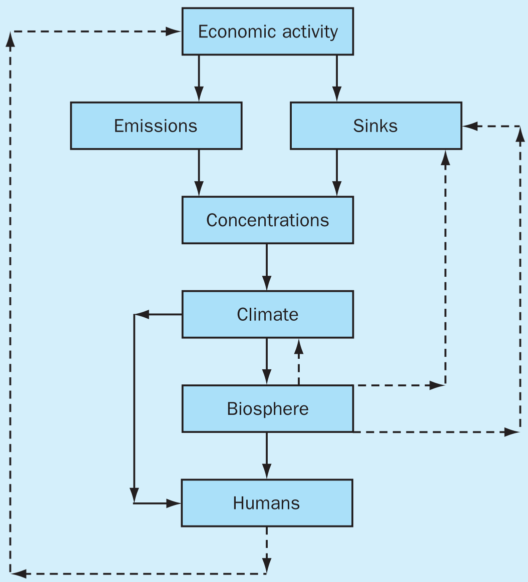


Figure 13.6 The enhanced greenhouse effect

to the problem. Article 2, Objectives, states that:

> The ultimate objective of this Convention and any related legal instruments that the Conference of the Parties may adopt is to achieve, in accordance with the relevant provisions of the Convention, stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system.

This could be seen as being in the nature of a modified SMS in that Article 2 goes on to say that this is subject to enabling 'economic development to proceed in a sustainable manner'.

The role of uncertainty is explicitly recognised in Article 3, Principles, where it is stated that

> 3. The Parties should take precautionary measures to anticipate, prevent or minimize the causes of climate change and mitigate its adverse effects. Where there are threats of serious or irreversible damage, lack of full scientific certainty should not be used as a reason for postponing such measures, . . .

Neither the 1992 treaty nor subsequent Conferences of the Parties have translated this presumption in favour of precaution into any kind of operational target corresponding to the objective, be that in terms of greenhouse gas concentrations or temperature change. The EU also adopts a precautionary approach, and it has operationalised it as meaning that the change in global average temperature should not be more than 2°C above the pre-industrial level. A number of NGOs have endorsed this objective.

The extent to which economists have taken on board imperfect knowledge of the future in defining an objective for the mitigation of anthropogenic climate change varies greatly. Thus far, it appears to have been most comprehensively dealt with in the Stern Review (Stern, 2006), which we discussed aspects of in Chapters 3 (see Box 3.1 on ethics), 9, and 11 (see Box 11.2 on the consumption discount rate).

Stern emphasises the roles of imperfect knowledge and irreversibility from its outset. At the start of the Executive Summary it states that:

> Our actions over the coming few decades could create the risk of major disruptions to economic and social activity, later in this century and in the next, on a scale similar to those associated with the great wars and the economic depression of the first half of the 20th century. And it will be difficult or impossible to reverse these changes.

In regard to the consequences of climate change on BAU, Stern states that these

> are now expected to be more serious than many earlier studies suggested, not least because those studies tended to exclude some of the most uncertain but potentially most damaging impacts . . .

In Chapter 2 Stern notes the distinction between risk and uncertainty in terms of the ability to assign probabilities, and that the latter suggests a precautionary principle based approach to decision making. It is also noted that some recent work has provided a 'formal description' for that principle. By this it is meant that where there are a number of equally likely probability distributions, it can be shown 'under formal but reasonable assumptions' the decision maker would act 'as if she chooses the action that maximises a weighted average of the worst expected utility and the best expected utility' where 'the weight placed on the worst outcome would be influenced by the concern of the individual about the magnitude of associated threats, or pessimism, and possibly any hunch about which probability might be more or less plausible' (p. 39, where references to the original

Box 13.1 ***continued***

work are provided). This 'expression of the "precautionary principle"' is 'different from and additional to the idea of "aversion to risk" associated with and derived from expected utility' (p. 39) – aversion to risk is what we called risk aversion in section 13.1.

While, one version of, the risk/uncertainty distinction is made, and the basis for a precautionary premium noted, in Chapter 2 of the review, in the actual analysis that supports its conclusions Stern uses the standard expected utility framework. The problem that is being dealt with is, however, generally referred to as 'uncertainty'. Given the utility function used, with diminishing marginal utility, this entails, as discussed in section 13.1 here, risk aversion. Stern notes that increases in greenhouse gas concentrations are 'irreversible in the short to medium run' and very difficult to reverse 'even in the ultra-long run' (p. 328), and that the way new information can be used is asymmetric, so that what we called quasi-option value, but Stern calls option value, applies. However, it is stated that this is not 'clear-cut' as a similar premium exists with respect to delaying investment in mitigation capital equipment. In the event, quasi-option value is not used in Stern's formal analysis.

Stern's treatment of the costs of climate change is covered in Chapter 6 of the review, and it is here that the expected utility framework is used. An integrated assessment model, IAM, is used to produce projections for global GDP under business as usual (BAU) with climate change, and for BAU with the climate changing parts of the model 'switched-off'. The difference gives a trajectory of GDP losses on account of climate change, which is converted to consumption and then to utility losses, and the result expressed in terms of BGE loss as discussed in Box 3.1 here. The IAM has many parameters for the economic and for the climate system, the values of which are imperfectly known. For each climatic parameter the Stern review team constructed a (subjective) probability distribution based on a review of the scientific literature. For a given scenario, the IAM was actually run 1000 times, and in each run each of the climatic parameters had its value set as a random selection from its probability distribution. The result is, for each scenario, 1000 different BGE losses. The outcomes are reported in Stern in terms of the mean across the 1000 runs, and the 5th and 95th percentiles.

Scenarios differed in two respects. First, in terms of the range of impacts included in the evaluation of BGE. As well as market impacts, some scenarios included 'risk of catastrophe', which means that the effects of possible catastrophic damage effects at higher temperatures were allowed for, as well as the more gradual effects included in the markets impacts scenario. A third scenario added in also direct, non-market, impacts on human health and the environment. Second, a set of 'High Climate' scenarios were run. These scenarios are intended to explore the 'the impacts that may be seen if the level of temperature change is pushed to higher levels through the action of amplifying feedbacks in the climate system' (p. 175). Figure 13.6 shows feedback paths from the impact of climate change on the biosphere to the climate for given concentrations (albedo effects for example), to concentrations, and to sinks. The last two paths are included in the High Climate scenarios as, respectively, resulting from increased natural methane releases with higher temperatures, and from weakened carbon sinks as plant and soil respiration increases with temperature.

Table 13.5 shows results from this exercise for the economic losses from climate change under BAU. In Box 3.1 we reported that Stern had the costs of BAU as equivalent to a reduction in global per capita consumption of 10.9% now and forever. You can now see that we were there referring to the mean estimate for baseline climate assumptions. Table 13.5 shows the spread around this. If only market impacts are considered, with baseline climate (no feedbacks) there is a 5% probability, according to Stern's IAM, that economic damage could be as low as 0.3%. For the same climate, and considering the full range of impacts, there is a 5% probability that it could be as high as 27.4%. If the two feedbacks are included, the mean goes to 14.4 for the full range of impacts, the 5% lower limit is unchanged, and the 5% upper limit goes to 32.6%.

Stern's account of the review's assessment of the costs of mitigation is in Chapters 9 and 10. Chapter 9 takes a 'bottom-up' approach, looking at individual technologies and estimates of the costs of switching to them. Chapter 10 takes a 'top-down' approach, looking at macroeconomic models. Neither chapter deals with imperfect knowledge in a formal way. In both cases, the broad results are expressed in terms of

Box 13.1 *continued*

Table 13.5 Losses on BAU across six scenarios

Scenario		BGE equivalents: % loss in current consumption		
Climate	Economic	5th percentile	Mean	95th percentile
Baseline Climate	Market impacts	0.3	2.1	5.9
	Market impacts + risk of catastrophes	0.6	5.0	12.3
	Market impacts + risk of catastrophes + non-market impacts	2.2	10.9	27.4
High Climate	Market impacts	0.3	2.5	7.5
	Market impacts + risk of catastrophes	0.9	6.9	16.5
	Market impacts + risk of catastrophes + non-market impacts	2.7	14.4	32.6

Note: Utility discount rate 0.01, elasticity of marginal utility of consumption 1.
Source: Stern (2006) Table 6.1

percentages of GDP, and the two sets of results are similar. For the bottom-up approach, Stern reports an average estimate of the cost of cutting to 75% of current levels by 2050 as approximately 1% of GDP, with a range of –1 to +3.5%. This is consistent with stabilisation at 550ppm CO_{2e}. For the top-down approach, Stern reports an annual cost 'likely to be around 1% of GDP by 2050 with a range of +/– 3%, reflecting uncertainties over the scale of mitigation required, the pace of technological innovation and the degree of policy flexibility' (p. 267) for an emissions trajectory leading to stabilisation at around 500–550 ppm CO_{2e}.

Damage and mitigation costs are brought together in Chapter 13 in a discussion of what Stern considers to be an appropriate policy objective for mitigation. Stern notes that the objective could be specified in terms of impacts, warming, greenhouse gas concentrations, cumulative greenhouse gas emissions, or the reduction in annual emissions by a specific date. Concentrations are selected on the grounds that the aggregate is just one variable, CO_{2e}, that progress can be measured reasonably quickly, and that, with some uncertainty, there are understood links to human activity and human impacts. While Stern notes that the standard economic approach (followed by some other analysts) would be to set the target by optimising, so that (suitably discounted) marginal costs and benefits of mitigation are equalised, this is not what it tries to do.

Rather, Stern relies on an assessment of the disaggregated impacts and costs as reported in the review, and uses the formal aggregated results that we have reported here in summary form to support that assessment, rather than trying to have them fix an 'optimum' target. That this is the way Stern reached its conclusions is emphasised in the responses made to its critics in the exchange of papers following publication of the review (references in Chapter 3 here). Its main conclusion regarding the global policy objective is that the 'stabilisation goal should lie within the range 450–550 ppm CO_{2e}' (p. 338). It is noted that '550 ppm CO_{2e} would be a dangerous place to be, with substantial risks of very unpleasant outcome' (p. 329), and that aiming below 450 ppm CO_{2e} 'would impose very high adjustment costs in the near term for relatively small gains and might not even be feasible' (p. 318).

Summarising its modelling work, Stern says:

> [it] suggests that, allowing for uncertainty, if the world stabilises at 550 ppm CO_{2e}, climate change impacts could have an effect equivalent to reducing consumption today and forever by about 1.1%. As Chapter 6 showed, this compares with around 11% in the corresponding 'business as usual' case – ten times as high. With stabilisation at 450 ppm CO_{2e}, the percentage loss would be reduced to 0.6%, so choosing the tougher goal 'buys' about 0.5% of consumption now and forever. Choosing 550 ppm instead of 650 ppm CO_{2e} 'buys' about 0.6%. As with all models, these numbers reflect heroic assumptions . . . They also entail explicit judgements about some of the ethical issues involved . . . all integrated assessment models are sensitive to the assumptions and they should be taken as only indicative of the quantitative impacts, given those assumptions. It should be noted that the results (on damage costs of BAU) quoted from Chapter 6 leave out much that is important, and the other models referred to there leave out more. (p. 334) (parenthesis added)

Box 13.1 *continued*

It would appear reasonable to summarise all of Stern's arguments as follows. A target of 550 ppm CO_{2e} is easily affordable, and would avoid the worst that can be envisaged (see Figure 13.4 in Stern for a schematic representation of non-monetarised impacts). A target of 450 ppm CO_{2e} would be safer, but would be a lot more costly, and may well not now be realistically attainable. If this is a fair summary, then it sounds quite similar to a story about a modified Safe Minimum Standard. In practice, an analysis developed in terms of risk and expected utility theory may lead to much the same conclusion as one developed in terms of uncertainty and decision theory.

As will be seen in Chapter 17, the standard economic approach to the question of the proper rate at which renewable resources should be harvested is based on intertemporal efficiency criteria. Basically, the harvest rate should be such that the rate of return on the stock of such a resource is equal to the rate of return on other forms of investment. However, this approach ignores unpredictable exogenous shocks that could, given harvesting, threaten the sustainability of the resource. In Chapter 17, we shall examine a proposal that the proper rate of renewable resource harvesting should be determined on a modified SMS basis, so as to avoid this threat.

13.6.1 Environmental performance bonds

The precautionary principle is closely related to the modified safe minimum standard and is gaining widespread acceptance, at the governmental and intergovernmental levels, as a concept that should inform environmental policy. Statements of the precautionary principle have been made by a number of governments, by individuals, and as part of international agreements. Thus, for example, Principle 15 of the June 1992 Rio Declaration is that:

> In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation.[6]

Like the SMS, the precautionary principle can be taken as saying that there is a presumption against going ahead with projects that have serious irreversible environmental consequences, unless it can be shown that not to go ahead would involve unacceptable costs. The question which arises is whether there are any policy instruments that are consistent with this approach to irreversibility and uncertainty, which could constitute a feasible means for its implementation in such a way as to avoid an outcome that simply prohibits such projects.

Environmental performance bonds have been suggested as a response to the problem of devising a means of project appraisal which takes on board the ideas behind the SMS and the precautionary principle. The basic ideas involved can be discussed by considering some firm which wishes to undertake a project involving major technological innovation, so that there is no past experience according to which probabilities can be assigned to all possible outcomes. Indeed, in so far as genuine novelty is involved, there is radical uncertainty in that not all of the possible outcomes can be anticipated. An example of such a project would have been the construction of the first nuclear power plant.

We assume that there is in existence an environmental protection agency (EPA) without permission from which the firm cannot go ahead with the project. The EPA takes independent expert advice on the project, and comes to a view about the worst conceivable environmental outcome of the project's going ahead. Approval of the project is then conditional on the firm depositing with the EPA ('posting') a bond of $\$x$, where this is the estimate of the social cost of the worst conceivable outcome. The bond is fully or partially returned to the firm at the end of the project's lifetime (defined by the longest-lasting

[6] The Rio Declaration (United Nations, 1993a) is a set of 27 short, non-binding, statements of principle unanimously adopted at the United Nations Conference on Environment and Development that took place in Rio de Janeiro in June 1992.

conceived consequence of the project, not by the date at which it ceases to produce output) according to the damage actually occurring over the lifetime. Thus, if there is no damage the firm gets back $\$x$, plus some proportion of the interest. The withheld proportion of the interest is to cover EPA administration costs and to finance EPA research. If the damage actually occurring is $\$y$, the firm gets back $\$x - \$y$, with appropriate interest adjustment. For $\$x$ equal to $\$y$, the firm gets nothing back, forfeiting the full value of the bond. It is, of course, possible that $\$y$ will turn out to be greater than $\$x$, in which case also the firm gets back \$0.

The advantages claimed for such an instrument are in terms of the incentives it creates for the firm to undertake research to investigate environmental impact and means to reduce it, as well as in terms of stopping projects. Taking the latter point first, suppose that the EPA decides on $\$x$ as the size of the bond, and that the firm assesses lifetime project net returns to it as $\$(x - 1)$, and accepts that $\$x$ is the appropriate estimate of actual damage to arise. Then it will not wish to go ahead with the project. If, however, the firm took the view that actual damage would be $\$(x - 2)$ or less, it would wish to go ahead with the project. The firm has, then, an incentive to itself assess the damage that the project could cause, and to research means to reduce that damage. Further, if it does undertake the project it has an ongoing incentive to seek damage-minimising methods of operation, so as to increase the eventual size of the sum returned to it, $\$x - \$y$. This incentive effect could be enhanced by having the size of the bond posted periodically adjustable. Thus, if the firm could at any point in time in the life of the project, on the basis of its research, convince the EPA that the worst conceivable lifetime damage was less than $\$x$, the original bond could be returned and a new one for an amount less than $\$x$ be posted.

At the end of the project's lifetime, the burden of proof as to the magnitude of actual damage would rest with the firm, not the EPA. The presumption would be, that is, that the bond was not returnable. It would be up to the firm to convince the EPA that actual damage was less than $\$x$ if it wished to get any of its money back. This would generate incentives for the firm to monitor damage in convincing ways, as well as to research means to minimise damage. In the event that damage up to the amount of the bond, $\$x$, occurred, society, as represented by the EPA, would have received compensation. If damage in excess of $\$x$ had occurred, society would not receive full compensation. Recall that $\$x$ is to be set at the largest amount of damage seen as conceivable by the EPA at the outset, on the basis of expert advice. A socially responsible EPA would have an incentive to take a cautious view of the available evidence, implying a high figure for $\$x$, so that society would not find itself uncompensated. This, it is argued, would coincide with the selfish motivations of EPA staff, since a higher $\$x$ would mean more funding available for EPA administration and research.

Environmental performance bonds are clearly an interesting idea for an addition to the range of instruments for environmental protection, given the pervasiveness of uncertainty and the need for research addressed to reducing it. In the form discussed here, they do not appear to be in use anywhere. Their usefulness would appear, as with other environmental policy instruments, to vary with particular circumstances, and clearly further consideration of the details of their possible implementation is warranted.

Summary

Risk and uncertainty

In a risky world probabilities can be assigned to all possible outcomes. Uncertainty is where this cannot be done.

Cost of risk bearing

Individuals are generally assumed to be risk averse, in which case there is a cost to risk that is the difference between expected value and certainty equivalent.

Option value

Option value is a risk aversion premium which exists when the decision maker is risk averse.

Quasi-option value

With irreversibility and the prospect of more information being available in the future, quasi-option value is the pay-off to deferring a decision until more information is available. Quasi-option value exists for a risk-neutral decision maker. It is the amount by which such a decision maker should reduce the expected value of development benefits before comparing them with costs.

Safe minimum standard

In an uncertain world, where very large irreversible loss is a possible consequence of a decision to go ahead with a project, the safe minimum standard entails assuming the loss to be so large that the minimax regret rule leads to the decision not to go ahead. The modified safe minimum standard says do not go ahead unless the social costs then entailed are unacceptably high.

The precautionary principle

Closely related to the modified safe minimum standard, the precautionary principle is a presumption against projects that could have serious irreversible consequences. It puts the 'burden of proof' on project proponents.

Further reading

The economic analysis of individual decision making in the face of risk is covered in standard intermediate and advanced microeconomics texts such as Deaton and Muellbauer (1980) or Kreps (1990).

Our treatment of option price and option value draws heavily on Ready (1995), which provides lots of references to the original literature, where important papers include Weisbrod (1964), Cicchetti and Freeman (1971), Bishop (1982), Freeman (1985), Plummer and Hartman (1986), Desvousges *et al.* (1987) and Boyle and Bishop (1987).

Arrow and Fisher (1974) introduced quasi-option value in the wilderness development context, on which see also Krutilla and Fisher (1975) and Fisher and Krutilla (1974). Henry (1974) independently published essentially the same arguments. Fisher and Hanemann (1986) discuss quasi-option value in the context of pollution with irreversible consequences, and illustrate its estimation. Graham-Tomasi (1995) provides a generalisation of the setting and arguments originally proposed by Arrow and Fisher.

Dixit and Pindyck (1994) consider project appraisal and show that an irreversible investment opportunity that need not be implemented immediately should only be undertaken now if its expected net present value exceeds the opportunity cost of keeping the investment option alive. The value of these options can be estimated using standard techniques for the valuation of call options developed in the financial markets literature. Coggins and Ramezani (1998) use those techniques to show explicitly that the value of the right to delay a decision equals Arrow and Fisher's quasi-option value.

Funtowicz and Ravetz (1991) consider uncertainty, as we define it, in the context of scientific knowledge about environmental problems, and argue that quality assurance requires the 'democratisation' of science. Faucheux and Froger (1995) consider decision making in the face of uncertainty in the context of policies for sustainable development. Dixit and Nalebuff (1991) is an excellent, and very easy-to-read, account of using game theory to make strategic choices in the context of uncertainty.

Young (2001) develops a critique of the use of expected utility for environmental decision making and argues that G.L.S. Shackle's model of decision making in the face of uncertainty, originally developed for analysing business behaviour, provides a superior explanation of how decision makers actually proceed, and is a better prescriptive model. The arguments are supported with a case study of the appraisal of a highway project in a developing country.

The safe minimum standard for conservation idea was originally proposed by Ciriacy-Wantrup (1968), and further developed by Bishop (1978). Barbier *et al*. (1994) review some contributions to the safe minimum standard and the precautionary principle in the context of biodiversity preservation; see also Chapters 11, 12 and 13 in Heywood (1995). Perrings (1991) considers the theoretical content of the precautionary principle in relation to problems of uncertain environmental impact of large spatial and temporal dimension. Cameron and Aboucher (1991) give another useful discussion of the precautionary principle. As described in the text here, environmental performance bonds appear to have first been proposed by Perrings (1989), and are discussed further in Costanza and Perrings (1990); see Shogren *et al*. (1993) for a critique of the Costanza and Perrings proposals.

A substantial literature now exists about pollution targets and instruments in a risky or uncertain world. References to some useful reading in this area were given in Part II of this book. The greenhouse effect, considered in Chapter 10, is an example *par excellence* of a pollution problem where all dimensions are subject to uncertainty. Risk and uncertainty also have a bearing, of course, on efficient and optimal resource depletion and harvesting, to be dealt with in Part IV of this book, where references to the literature will be provided.

Discussion questions

1. Is the loss of a species of plant or animal necessarily of economic concern? Is this true for every species that currently exists? Do we now suffer as a consequence of earlier extinctions?
2. How could the value of an environmental performance bond be set?
3. Should the safe minimum standard approach be applied to setting standards for environmental pollution? If so, how could it be done?

Problems

1. Consider an individual for whom Y is initially £100 and $U(Y) = Y^a$, offered a bet on the toss of a fair coin at a price of £5. For each of the pay-outs A and B, calculate the expected value of the Y outcome, the individual's expected utility, certainty equivalent and cost of risk bearing, for a taking the values 0.9, 0.95, 0.99. 0.999 and 1.0. In situation A, the individual gets £15 if he or she calls the way the coin falls correctly, and nothing if he or she gets it wrong. In B, the pay-out on a correct call is £10. Note that A is actuarially a very good bet, while B is actuarially fair, and identify the circumstances in which the individual would take the bet. Note also that from equation 13.4 the certainty equivalent is expected utility raised to the power $1/a$.

2. In Figure 13.4 with $\text{MNB}_1 = 10 - (A_1/2)$ and $\text{MNB}_2 = 20 - (A_2/2)$ known with certainty, find the levels of A_1 and A_2, (a) if there is no irreversibility, (b) if irreversibility applies but is ignored in decision making, and (c) if irreversibility is taken into account. Hence calculate the cost of ignoring irreversibility. Suppose now that there is imperfect knowledge of the future, and a risk-neutral decision-maker aware of and taking into account irreversibility assigns equal probabilities to the mutually exclusive future states where

$MNB_2 = 10 - (A_1/2)$ and $MNB_2 = 20 - (A_2/2)$. What will be the selected levels for A_1 and A_2?

3. The construction of a hydroelectric plant in a wilderness valley is under consideration. It is known that the valley contains an insect species found nowhere else, and the project includes relocating the insects. It is not known whether they can be successfully located. The pay-off matrix is

		State of nature	
		F	**A**
Decision	**P**	+70	−20
	A	+20	+20

where F and A stand for favourable and unfavourable, P is the decision to go ahead with the hydroelectric plant, A is the decision to proceed instead with a coal-fired plant, and the cell entries are NPV millions of £s. Favourable is the state of nature where species relocation is successful, unfavourable is where it is not. Ascertain the decisions following from adopting: (a) the principle of insufficient reason, (b) the maximin rule and (c) the maximax rule. Derive the regret matrix and ascertain the implications of the minimax regret rule, and compare the outcome with that arising from the safe minimum standard approach.

PART IV Natural resource exploitation

CHAPTER 14 The efficient and optimal use of natural resources

The Golden Rule is that there are no golden rules.

George Bernard Shaw, Maxims for Revolutionists, in *Man and Superman*

Learning objectives

Having read this chapter, the reader should be able to

- understand the ideas of 'efficient' and 'optimal' allocations of environmental resources
- recognise the relationship between – but also the difference between – the concepts of efficiency and optimality
- understand how questions relating to efficient and optimal use of environmental resources over time can be analysed using a class of models known as 'optimal growth models'
- appreciate the ways in which resource use patterns are linked with sustainability

Introduction

In this chapter, we construct a framework to analyse the use of natural resources over time. This will provide the basis for our investigations of non-renewable resource depletion and the harvesting of renewable resources that follow in Chapters 15 to 18. Our objectives in the present chapter are:

- to develop a simple economic model, built around a production function in which natural resources are inputs into the production process;
- to identify the conditions that must be satisfied by an economically efficient pattern of natural resource use over time;
- to establish the characteristics of a socially optimal pattern of resource use over time in the special case of a utilitarian social welfare function.

We shall be constructing a stylised model of the economy in order to address questions about the use of resources. Although the economics of our model are straightforward, some mathematics is required to analyse the model. To keep technical difficulties to a minimum, the main body of text avoids the use of mathematical derivations. It shows the logic behind, and the economic interpretations of, important results. Derivations of results are presented separately in Appendices 14.1, 14.2 and 14.3. It is not vital to read these appendices to follow the arguments in the chapter, but we strongly recommend that you do read them. The derivations are explained thoroughly. Appendix 14.1 is of particular importance as it takes the reader through a key mathematical technique, dynamic optimisation using the maximum principle. You are also urged to read Appendices 14.2 and 14.3 to see how the results discussed in the text are obtained.

PART I A SIMPLE OPTIMAL RESOURCE DEPLETION MODEL

14.1 The economy and its production function

We begin by specifying the model used in this chapter. The economy produces a single good, Q, which can be either consumed or invested. Consumption increases current well-being, while investment increases the capital stock, permitting greater consumption in the future. Output is generated through a production function using a single 'composite' non-renewable resource input, R. Beginning in this way, with just one type of natural resource, abstracts from any substitution effects that might take place between different kinds of natural resource. In Chapter 15, we shall see how our conclusions alter when more than one type of natural resource enters the production function.

In addition to the non-renewable resource, a second input – manufactured capital, K – enters the production function, which is written as

$$Q = Q(K, R) \tag{14.1}$$

This states that output has some functional relationship to the quantities of the two inputs that are used, but it does not tell us anything about the particular form of this relationship.[1] One possible type of production technology is the Cobb–Douglas (CD) form, consisting of the class of functions

$$Q = AK^{\alpha}R^{\beta} \tag{14.2}$$

where A, α and $\beta > 0$. An alternative form, widely used in empirical analysis, is the constant elasticity of substitution (CES) type, which comprises the family of functional forms

$$Q = A(\alpha K^{-\theta} + \beta R^{-\theta})^{-\varepsilon/\theta} \tag{14.3}$$

where A, ε, α, $\beta > 0$, $(\alpha + \beta) = 1$, and $-1 < \theta \neq 0$.

The CD and CES forms of production function do not exhaust all possibilities. In this chapter, we shall not be making any assumption as to which type of production function best represents the production technology of an economy, but rather work with a general form that might be CD, might be CES, or might be some other. Which functional form is the 'correct' one is an empirical question, and cannot be answered by theoretical argument alone.

14.2 Is the natural resource essential?

The characteristics of an optimal resource depletion path through time will be influenced by whether the natural resource is 'essential'. Essentialness of a resource could mean several things. First, a resource might be essential as a waste disposal and reprocessing agent. Given the ubiquitous nature of waste and the magnitude of the damages that waste can cause, resources do appear to be necessary as waste-processing agents. A resource might also be essential for human psychic satisfaction. Humans appear to need solitude and the aesthetic enjoyment derived from observing or being in natural environments. Thirdly, some resource might be ecologically essential in the sense that some or all of a relevant ecosystem cannot survive in its absence.

In this chapter, we are concerned with a fourth meaning: whether a resource is directly essential for production. Some resources are undoubtedly essential for specific products – for example, crude oil is an essential raw material for the production of petrol and paraffin. But here we are conceptualising resources at a high degree of aggregation, dealing with general classes such as non-renewable and renewable resources. A productive input is defined to be essential if output is zero whenever the quantity of that input is zero, irrespective of the amounts of other inputs used. That is, R is essential if $Q = Q(K, R = 0) = 0$ for any positive value of K.

In the case of the CD production function, R and K are both essential, as setting any input to zero in equation 14.2 results in $Q = 0$. Matters are not so

[1] Each output level Q satisfying the production function is the maximum attainable output for given quantities of the inputs, and implies that inputs are used in a technically efficient way. The production function does not contain labour as a productive input; we have omitted labour to keep the algebra as simple as possible. One could choose to interpret K and R as being in per capita units, so that labour does implicitly enter as a productive input.

straightforward with the CES function. We state (but without giving a proof) that if $\theta < 0$ then no input is essential, and if $\theta > 0$ then all inputs are essential.

What is the relevance of this to our study of resource use over time? If we wish to answer questions about the long-run properties of economic systems, the essentialness of non-renewable resources will matter. Since, by definition, non-renewable resources exist in finite quantities it is not possible to use constant and positive amounts of them over infinite horizons. However, if a resource is essential, then we know that production can only be undertaken if some positive amount of the input is used. This seems to suggest that production and consumption cannot be sustained indefinitely if a non-renewable resource is a necessary input to production.

However, if the rate at which the resource is used were to decline asymptotically to zero, and so never actually become zero in finite time, then production could be sustained indefinitely even if the resource were essential. Whether output could rise, or at least stay constant over time, or whether it would have to decline towards zero will depend upon the extent to which other resources can be substituted for non-renewable resources and upon the behaviour of output as this substitution takes place.

14.3 What is the elasticity of substitution between *K* and *R*?

The extent of substitution possibilities is likely to have an important bearing on the feasibility of continuing economic growth over the very long run, given the constraints which are imposed by the natural environment. Let us examine substitution between the non-renewable resource and capital. The elasticity of substitution, σ, between capital and the non-renewable natural resource (from now on just called the resource) is defined as the proportionate change in the ratio of capital to the resource in response to a proportionate change in the ratio of the marginal products of capital and the resource, conditional on total output Q remaining constant (see Chiang, 1984). That is,

$$\sigma = \frac{d(K/R)}{K/R} \bigg/ \frac{d(Q_R/Q_K)}{Q_R/Q_K} \mid Q = \text{constant} \qquad (14.4)$$

where the partial derivative $Q_R = \partial Q/\partial R$ denotes the marginal product of the resource and $Q_K = \partial Q/\partial K$ denotes the marginal product of capital.[2] The elasticity of substitution lies between zero and infinity. Substitution possibilities can be represented diagrammatically. Figure 14.1 shows what are known as production function isoquants. For a given production function, an isoquant is the locus of all combinations of inputs which, when used efficiently, yield a constant level of output. The three isoquants shown in Figure 14.1 each correspond to the level of output, $\bar{Q}$, but derive from different production

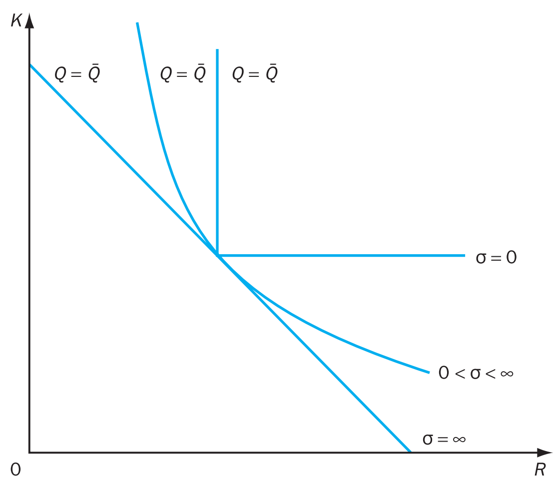


Figure 14.1 Substitution possibilities and the shapes of production function isoquants

[2] It can also be shown (see Chiang, 1984, for example) that if resources are allocated efficiently in a competitive market economy, the elasticity of substitution between capital and a non-renewable resource is equal to

$$\sigma = \frac{d(K/R)}{K/R} \bigg/ \frac{d(P_R/P_K)}{P_R/P_K}$$

where P_R and P_K denote the unit prices of the non-renewable resource and capital, respectively. That is, the elasticity of substitution measures the proportionate change in the ratio of capital to non-renewable resource used in response to a change in the relative price of the resource to capital.

functions. The differing substitution possibilities are reflected in the curvatures of the isoquants.

In the case where no input substitution is possible (that is, $\sigma = 0$), inputs must be combined in fixed proportions and the isoquants will be L-shaped. Production functions of this type, admitting no substitution possibilities, are sometimes known as Leontief functions. They are commonly used in input–output models of the economy. At the other extreme, if substitution is perfect ($\sigma = \infty$), isoquants will be straight lines. In general, a production function will exhibit an elasticity of substitution somewhere between those two extremes (although not all production functions will have a constant σ for all input combinations). In these cases, isoquants will often be convex to the origin, exhibiting a greater degree of curvature the lower the elasticity of substitution, σ. Some evidence on empirically observed values of the elasticity of substitution between different production inputs is presented later in Box 14.1.

For a CES production function, we can also relate the elasticity of substitution to the concept of essentialness. It can be shown (see, for example, Chiang, 1984, p. 428) that $\sigma = 1/(1 + \theta)$. We argued in the previous section that no input is essential where $\theta < 0$, and all inputs are essential where $\theta > 0$. Given the relationship between σ and θ, it can be seen that no input is essential where $\sigma > 1$, and all inputs are essential where $\sigma < 1$. Where $\sigma = 1$ (that is, $\theta = 0$), the CES production function collapses to the CD form, where all inputs are essential.

14.4 Resource substitutability and the consequences of increasing resource scarcity

As production continues through time, stocks of non-renewable resources must decline. Continuing depletion of the resource stock will lead to the non-renewable resource price rising relative to the price of capital. As the relative price of the non-renewable resource rises the resource to capital ratio will fall, thereby raising the marginal product of the resource and reducing the marginal product of capital. However, the magnitude of this substitution effect will depend on the size of the elasticity of substitution. Where the elasticity of substitution is high, only small changes in relative input prices will be necessary to induce a large proportionate change in the quantities of inputs used. 'Resource scarcity' will be of little consequence as the economy is able to replace the scarce resource by the reproducible substitute. Put another way, the constraints imposed by the finiteness of the non-renewable resource stock will bite rather weakly in such a case.

On the other hand, low substitution possibilities mean that as resource depletion pushes up the relative price of the resource, the magnitude of the induced substitution effect will be small. 'Resource scarcity' will have more serious adverse effects, as the scope for replacement of the scarce resource by the reproducible substitute is more limited. Where the elasticity of substitution is zero, then no scope exists for such replacement.

14.4.1 The feasibility of sustainable development

In Chapter 3, we considered what sustainability might mean, how economists have attempted to incorporate a concern with sustainability into their work, and why one might wish to incorporate sustainability into the set of objectives that society pursues. We begin here by considering whether sustainable development is actually possible.

To address this question, two things are necessary. First, a criterion of sustainability is required; unless we know what sustainability is it is not possible to judge whether it is feasible. Second, we need to describe the material transformation conditions available to society, now and in the future. These conditions – the economy's production possibilities – determine what can be obtained from the endowments of natural and human-made capital over the relevant time horizon.

To make some headway in addressing this question a conventional sustainability criterion will be adopted: non-declining per capita consumption maintained over indefinite time (see Chapter 3). Turning attention to the transformation conditions, it is clear that a large number of factors enter the picture. What is happening to the size of the human population? What kinds of resources are available and in what quantities, and what properties do they possess? What will happen to the state of technology in the future? How will ecosystems be affected by the continuing waste loads being placed upon the environment, and how will ecosystem changes feed back upon productive potential? To make progress, economists typically simplify and narrow down the scope of the problem, and represent the transformation possibilities by making an assumption about the form of an economy's production function. A series of results have become established for several special cases, deriving mainly from papers by Dasgupta and Heal (1974), Solow (1974a) and Stiglitz (1974). For the CD and CES functions we have the following.

Case A: Output is produced under fully competitive conditions through a CD production function with constant returns to scale and two inputs, a non-renewable resource, R, and manufactured capital, K, as in the following special case of equation 14.2:

$$Q = K^{\alpha}R^{\beta} \text{ with } (\alpha + \beta) = 1$$

Then, in the absence of technical progress and with constant population, it is feasible to have constant consumption across generations if the share of total output going to capital is greater than the share going to the natural resource (that is, if $\alpha > \beta$).

Case B: Output is produced under fully competitive conditions through a CES production function with constant returns to scale and two inputs, a non-renewable resource, R, and manufactured capital, K, as in equation 14.3:

$$Q = A(\alpha K^{-\theta} + \beta R^{-\theta})^{-\varepsilon/\theta} \text{ with } (\alpha + \beta) = 1$$

Then, in the absence of technical progress and with constant population, it is feasible to have constant consumption across generations if the elasticity of substitution $\sigma = 1/(1 + \theta)$ is greater than or equal to one.

Case C: Output is produced under conditions in which a backstop technology is permanently available. In this case, the non-renewable natural resource is not essential. Sustainability is feasible, although there may be limits to the size of the constant consumption level that can be obtained.

It is relatively easy to gain some intuitive understanding of these results. For the CD case, although the natural resource is always essential in the sense we described above, if $\alpha > \beta$ then capital is sufficiently substitutable for the natural resource so that output can be maintained by increasing capital as the depletable resource input diminishes. However, it should be noted that there is an upper bound on the amount of output that can be indefinitely sustained in this case; whether that level is acceptably high is another matter. For the CES case, if $\sigma > 1$, then the resource is not essential. Output can be produced even in the absence of the natural resource. The fact that the natural resource is finite does not prevent indefinite production (and consumption) of a constant, positive output. Where $\sigma = 1$, the CES production function collapses to the special case of CD, and so Case A applies. Where a backstop exists (such as a renewable energy source like wind or solar power, or perhaps nuclear-fusion-based power) then it is always possible to switch to that source if the limited natural resource becomes depleted. We explore this process further in the next chapter.

These results assumed that the rate of technical progress and the rate of population growth were both zero. Not surprisingly, results change if one or both of these rates is non-zero. The presence of permanent technical progress increases the range of circumstances in which indefinitely long-lived constant per capita consumption is feasible whereas constant population growth has the opposite effect. However, there are circumstances in which constant per capita consumption can be maintained even where population is growing provided the rate of technical progress is sufficiently large and the share of output going to the resource is sufficiently low. Details of this result are given in Stiglitz (1974). Similarly, for a CES production function, sustained consumption is possible even where $\sigma < 1$ provided that technology growth is sufficiently high relative to population growth.

The general conclusion from this analysis is that the feasibility of sustainability requires either a relatively high degree of substitutability between capital and the resource, or a sufficiently large continuing rate of technical progress or the presence of a permanent backstop technology. Whether such conditions will prevail is a moot point.

14.4.2 Sustainability and the Hartwick rule

In our discussion of sustainability in Chapter 3, mention was made of the so-called Hartwick savings rule. Interpreting sustainability in terms of non-declining consumption through time, John Hartwick (1977, 1978) sought to identify conditions for sustainability. He identified two sets of conditions which were sufficient to achieve constant (or, more accurately, non-declining) consumption through time:

- a particular savings rule, known as the Hartwick rule, which states that the rents derived from an efficient extraction programme for the non-renewable resource are invested entirely in reproducible (that is, physical and human) capital;
- conditions pertaining to the economy's production technology. These conditions are essentially those we described in the previous section, which we shall not repeat here.

We shall discuss the implications of the Hartwick rule further in Chapter 19. But three comments about it are worth making at this point. First, the Hartwick rule is essentially an *ex post* description of a sustainable path. Hence if an economy were not *already* on a sustainable path, then adopting the Hartwick rule is not sufficient for sustainability from that point forwards. This severely reduces the practical usefulness of the 'rule'. (See Asheim, 1986, and Pezzey, 1996, and Appendix 19.1 in the present book.) Second, even were the economy already on a sustainable path, the Hartwick rule requires that the rents be generated from an *efficient* resource extraction programme in a competitive economy. Third, even if the Hartwick rule is pursued subject to this qualification, the savings rule itself does not guarantee sustainability. Technology conditions may rule out the existence of a feasible path. As we noted in the previous section, feasibility depends very much upon the extent of substitution possibilities open to an economy. Let us now explore this a little further.

14.4.3 How large are resource substitution possibilities?

Clearly, the magnitude of the elasticity of substitution between non-renewable resources and other inputs is a matter of considerable importance. But how large it is cannot be deduced by *a priori* theoretical reasoning – this magnitude has to be inferred empirically. Whereas many economists believe that evidence points to reasonably high substitution possibilities (although there is by no means a consensus on this), natural scientists and ecologists stress the limited substitution possibilities between resources and reproducible capital. Indeed some ecologists have argued that, in the long term, these substitution possibilities are zero.

These disagreements reflect, in large part, differences in conceptions about the scope of services that natural resources provide. For example, whereas it appears to be quite easy to economise on the use of fossil energy inputs in many production processes, reproducible capital cannot substitute for natural capital in the provision of the amenities offered by wilderness areas, or in the regulation of the earth's climate. The reprocessing of harmful wastes is less clear-cut; certainly reproducible capital and labour can substitute for the waste disposal functions of the environment to some extent (perhaps through increased use of recycling processes) but there appear to be limits to how far this substitution can proceed.

Finally, it is clear that even if we were to establish that substitutability had been high in the past, this does not imply that it will continue to be so in the future. It may be that as development pushes the economy to points where natural constraints begin to bite, substitution possibilities reduce significantly. Recent literature from natural science seems to suggest this possibility. On the other hand, a more optimistic view is suggested by the effect of technological progress, which appears in many cases to have contributed towards enhanced opportunities for substitution. You should now read the material on resource substitutability presented in Box 14.1.

Up to this point in our presentation, natural resources have been treated in a very special way. We have assumed that there is a single, non-renewable resource, *R*, of fixed, known size, and (implicitly) of uniform quality. Substitution possibilities have been limited to those between this resource and other, non-natural, resources. In practice, there are a large number of different natural resources, with substitution possibilities between members of this set. Of equal importance is the non-uniform quality of

Box 14.1 Resource substitutability: one item of evidence

A huge amount of empirical research has been devoted to attempts to measure the elasticity of substitution between particular pairs of inputs. Results of these exercises are often difficult to apply to general models of the type we use in this chapter, because the estimates tend to be specific to the particular contexts being studied, and because many studies work at a much more disaggregated level than is done here.

We restrict comments to just one estimate, which has been used in a much-respected model of energy–environment interactions in the United States economy.

Alan Manne (who unfortunately died in 2005), in developing the ETA Macro model, considered a production function in which gross output (Q) depends upon four inputs: K, L, E and N (respectively capital, labour, electric and non-electric energy). Manne's production function incorporates the following assumptions:

(a) There are constant returns to scale in terms of all four inputs.
(b) There is a unit elasticity of substitution between capital and labour.
(c) There is a unit elasticity of substitution between electric and non-electric energy.
(d) There is a constant elasticity of substitution between the two pairs of inputs, capital and labour on the one hand and electric and non-electric energy on the other. Denoting this constant elasticity of substitution by the symbol σ, the production function used in the ETA Macro model that embraces these assumptions is

$$Q = [a(K^{\alpha}L^{1-\alpha})^{-\theta} + b(E^{\beta}N^{1-\beta})^{-\theta}]^{-1/\theta}$$

where, as noted in the text,

$$\sigma = \frac{1}{1+\theta}$$

Manne selects the value 0.25, a relatively low figure, for the elasticity of substitution σ between the pair of energy inputs and the other input pair. How is this figure arrived at? First, Manne argues that the elasticity of substitution is approximately equal to the absolute value of the price elasticity of demand for primary energy (see Hogan and Manne, 1979). Then, Manne collects time-series data on the prices of primary energy, incomes and quantities of primary energy consumed. This permits a statistically derived estimate of the long-run price elasticity of demand for primary energy to be obtained, thereby giving an approximation to the elasticity of substitution between energy and other production inputs. Manne's elasticity estimate of 0.25 falls near the median of recent econometric estimates of this elasticity of substitution.

Being positive, this figure suggests that energy demand will rise relative to other input demands if the relative price of other inputs to energy rises, and so the composite energy resource is a substitute for other productive inputs (a negative sign would imply the pair were complements). However, as the absolute value of the elasticity is much less than one, the degree of substitutability is very low, implying that relative input demands will not change greatly as relative input prices change.

Source: Manne (1979)

resource stocks. Resource stocks do not usually exist in a fixed amount of uniform quality, but rather in deposits of varying grade and quality. As high-grade reserves become exhausted, extraction will turn to lower-grade deposits, provided the resource price is sufficiently high to cover the higher extraction costs of the lower-grade mineral. Furthermore, while there will be some upper limit to the physical occurrence of the resource in the earth's crust, the location and extent of these deposits will not be known with certainty. As known reserves become depleted, exploration can, therefore, increase the size of available reserves. Finally, renewable resources can act as backstops for non-renewable: wind and wave power are substitutes for fossil fuels, and wood products are substitutes for metals for some construction purposes, for example.

Dasgupta (1993) examines these various substitution possibilities. He argues that they can be classified into nine innovative mechanisms:

1. an innovation allowing a given resource to be used for a given purpose. An example is the use of coal in refining pig-iron;
2. the development of new materials, such as synthetic fibres;

3. technological developments which increase the productivity of extraction processes. For example, the use of large-scale earthmoving equipment facilitating economically viable strip-mining of low-grade mineral deposits;
4. scientific and technical discovery which makes exploration activities cheaper. Examples include developments in aerial photography and seismology;
5. technological developments that increase efficiency in the use of resources. Dasgupta illustrates this for the case of electricity generation: between 1900 and the 1970s, the weight of coal required to produce one kilowatt-hour of electricity fell from 7 lb to less than 1 lb;
6. development of techniques which enable one to exploit low-grade but abundantly available deposits. For example, the use of froth-flotation, allowing low-grade sulphide ores to be concentrated in an economical manner;
7. constant developments in recycling techniques which lower costs and so raise effective resource stocks;
8. substitution of low-grade resource reserves for vanishing high-grade deposits;
9. substitution of fixed manufacturing capital for vanishing resources.

In his assessment of substitution possibilities, Dasgupta (p. 1126) argues that only one of these nine mechanisms is of limited scope, the substitution of fixed manufacturing capital for natural resources:

> Such possibilities are limited. Beyond a point fixed capital in production is complementary to resources, most especially energy resources. Asymptotically, the elasticity of substitution is less than one.

There is a constant tension between forces which raise extraction and refining costs – the depletion of high-grade deposits – and those which lower such costs – discoveries of newer technological processes and materials. What implications does this carry for resource scarcity? Dasgupta argues that as the existing resource base is depleted, profit opportunities arise from expanding that resource base; the expansion is achieved by one or more of the nine mechanisms just described. Finally, in a survey of the current stocks of mineral resources, Dasgupta notes that after taking account of these substitution mechanisms, and assuming unchanged resource stock to demand ratios:

> the only cause for worry are the phosphates (a mere 1300 years of supply), fossil fuels (some 2500 years), and manganese (about 130 000 years). The rest are available for more than a million years, which is pretty much like being inexhaustible.

However, adjusting for population and income growth,

> the supply of hydrocarbons . . . will only last a few hundred years . . . So then, this is the fly in the ointment, the bottleneck, the binding constraint.

Dasgupta's optimism is not yet finished. He conjectures that profit potentials will induce technological advances (perhaps based on nuclear energy, perhaps on renewables) that will overcome this binding constraint. Not all commentators share this sanguine view, as we have seen previously, and we shall have more to say about resource scarcity in the next chapter. In the meantime, we return to our simple model of the economy, in which the heterogeneity of resources is abstracted from, and in which we conceive of there being one single, uniform, natural resource stock.

14.5 The social welfare function and an optimal allocation of natural resources

Chapters 5 and 11 established the meaning of the concepts of efficiency and optimality for the allocation of productive resources in general. We shall now apply these concepts to the particular case of natural resources. Our objective is to establish what conditions must be satisfied for natural resource allocation to be optimal, in the sense that the allocation maximises a social welfare function. The presentation in this chapter focuses upon non-renewable resources, although we also indicate how the ideas can be applied to renewable resources.

The first thing we require is a social welfare function. You already know that a general way of writing the social welfare function (SWF) is:

$$W = W(U_0, U_1, U_2, \ldots, U_T) \tag{14.5}$$

where U_t, $t = 0, \ldots, T$, is the aggregate utility in period t.[3] We now assume that the SWF is utilitarian in form. A utilitarian SWF defines social welfare as a weighted sum of the utilities of the relevant individuals. As we are concerned here with inter-temporal welfare, we can interpret an 'individual' to mean an aggregate of persons living at a certain point in time, and so refer to the utility in period 0, in period 1, and so on. Then a utilitarian inter-temporal SWF will be of the form

$$W = \alpha_0 U_0 + \alpha_1 U_1 + \alpha_2 U_2 + \ldots + \alpha_T U_T \qquad (14.6)$$

Now let us assume that utility in each period is a concave function of the level of consumption in that period, so that $U_t = U(C_t)$ for all t, with $U_C > 0$ and $U_{CC} < 0$. Notice that the utility function itself is not dependent upon time, so that the relationship between consumption and utility is the same in all periods. By interpreting the weights in equation 14.6 as discount factors, related to a social utility discount rate ρ that we take to be fixed over time, the social welfare function can be rewritten as

$$W = U_0 + \frac{U_1}{1+\rho} + \frac{U_2}{(1+\rho)^2} + \ldots + \frac{U_T}{(1+\rho)^T} \qquad (14.7)$$

For reasons of mathematical convenience, we switch from discrete-time to continuous-time notation, and assume that the relevant time horizon is infinite. This leads to the following special case of the utilitarian SWF:

$$W = \int_{t=0}^{t=\infty} U(C_t)\mathrm{e}^{-\rho t}\mathrm{d}t \qquad (14.8)$$

There are two constraints that must be satisfied by any optimal solution. First, all of the resource stock is to be extracted and used by the end of the time horizon (as, after this, any remaining stock has no effect on social well-being). Given this, together with the fact that we are considering a non-renewable resource for which there is a fixed and finite initial stock, the total use of the resource over time is constrained to be equal to the fixed initial stock. Denoting the initial stock (at $t = 0$) as S_0 and the rate of extraction and use of the resource at time t as R_t, we can write this constraint as

$$S_t = S_0 - \int_{\tau=0}^{\tau=t} R_\tau \,\mathrm{d}\tau \qquad (14.9)$$

Notice that in equation 14.9, as we are integrating over a time interval from period 0 to any later point in time t, it is necessary to use another symbol (here τ, the Greek letter tau) to denote any point in time in the range over which the function is being integrated. Equation 14.9 states that the stock remaining at time t (S_t) is equal to the magnitude of the initial stock (S_0) less the amount of the resource extracted over the time interval from zero to t (given by the integral term on the right-hand side of the equation). An equivalent way of writing this resource stock constraint is obtained by differentiating equation 14.9 with respect to time, giving

$$\dot{S}_t = -R_t \qquad (14.10)$$

where the dot over a variable indicates a time derivative, so that $\dot{S}_t = \mathrm{d}S/\mathrm{d}t$. Equation 14.10 has a straightforward interpretation: the rate of depletion of the stock, $-\dot{S}_t$, is equal to the rate of resource stock extraction, R_t.

A second constraint on welfare optimisation derives from the accounting identity relating consumption, output and the change in the economy's stock of capital. Output is shared between consumption goods and capital goods, and so that part of the economy's output which is not consumed results in a capital stock change. Writing this identity in continuous-time form we have[4]

$$\dot{K}_t = Q_t - C_t \qquad (14.11)$$

[3] Writing the SWF in this form assumes that it is meaningful to refer to an aggregate level of utility for all individuals in each period. Then social welfare is a function of these aggregates, but not of the distribution of utilities between individuals within each time period. That is a very strong assumption, and by no means the only one we might wish to make. We might justify this by assuming that, for each time period, utility is distributed in an optimal way between individuals.

[4] Notice that by integration of equation 14.11 we obtain

$$K_t = K_0 + \int_{\tau=0}^{\tau=t} (Q_\tau - C_\tau)\mathrm{d}\tau$$

in which K_0 is the initial capital stock (at time zero). This expression is equivalent in form to equation 14.9 in the text.

It is now necessary to specify how output, Q, is determined. Output is produced through a production function involving two inputs, capital and a non-renewable resource:

$$Q_t = Q(K_t, R_t) \tag{14.12}$$

Substituting for Q_t in equation 14.11 from the production function 14.12, the accounting identity can be written as

$$\dot{K}_t = Q(K_t, R_t) - C_t \tag{14.13}$$

We are now ready to find the solution for the socially optimal intertemporal allocation of the non-renewable resource. To do so, we need to solve a constrained optimisation problem. The objective is to maximise the economy's social welfare function subject to the non-renewable resource stock–flow constraint and the national income identity. Writing this mathematically, therefore, we have the following problem:

Select values for the choice variables C_t and R_t for $t = 0, \ldots, \infty$ so as to maximize

$$W = \int_{t=0}^{t=\infty} U(C_t)\mathrm{e}^{-\rho t}\mathrm{d}t$$

subject to the constraints

$$\dot{S}_t = -R_t$$

and

$$\dot{K}_t = Q(K_t, R_t) - C_t$$

The full solution to this constrained optimisation problem, and its derivation, are presented in Appendix 14.2. This solution is obtained using the maximum principle of optimal control. That technique is explained in Appendix 14.1, which you are recommended to read now. Having done that, then read Appendix 14.2, where we show how the maximum principle is used to solve the problem that has been posed in the text. If you find this appendix material difficult, note that the text of this chapter has been written so that it can be followed without having read the appendices. In the following sections, we outline the nature of the solution, and provide economic interpretations of the results.

14.5.1 The nature of the solution

Four equations characterise the optimal solution:

$$U_{C,t} = \omega_t \tag{14.14a}$$

$$P_t = \omega_t Q_{R,t} \tag{14.14b}$$

$$\dot{P}_t = \rho P_t \tag{14.14c}$$

$$\dot{\omega}_t = \rho\omega_t - Q_{K,t}\omega_t \tag{14.14d}$$

Before we discuss the economic interpretations of these equations, it is necessary to explain several things about the notation used and the nature of the solution:

- The terms Q_K $(= \partial Q/\partial K)$ and Q_R $(= \partial Q/\partial R)$ are the partial derivatives of output with respect to capital and the non-renewable resource. In economic terms, they are the marginal products of capital and the resource, respectively. Time subscripts are attached to these marginal products to make explicit the fact that their values will vary over time in the optimal solution.
- The terms P_t and ω_t are the shadow prices of the two productive inputs, the natural resource and capital. These two variables carry time subscripts because the shadow prices will vary over time. The solution values of P_t and ω_t, for $t = 0, 1, \ldots, \infty$, define optimal time paths for the prices of the natural resource and capital.[5]
- The quantity being maximised in equation 14.8 is a sum of (discounted) units of utility. Hence the shadow prices are measured in utility, not consumption (or money income), units. You should now turn to Box 14.2 where an explanation of the relationship between prices in utils and prices in consumption (or income) units is given.

[5] A shadow price is a price that emerges as a solution to an optimisation problem; put another way, it is an implicit or 'planning' price that a good (or in this case, a productive input) will take if resources are allocated optimally over time. If an economic planner were using the price mechanism to allocate resources over time, then $\{P_t\}$ and $\{\omega_t\}$, $t = 0, 1, \ldots, \infty$, would be the prices he or she should establish in order to achieve an efficient and optimal resource allocation.

Box 14.2 Prices in units of utility: what does this mean?

The notion of prices being measured in units of utility appears at first sight a little strange. After all, we are used to thinking of prices in units of money: a Cadillac costs \$40 000, a Mars bar 30 pence, and so on. Money is a claim over goods and services: the more money someone has, the more goods he or she can consume. So it is evident that we could just as well describe prices in terms of consumption units as in terms of money units. For example, if the price of a pair of Levi 501 jeans were \$40, and we agree to use that brand of jeans as our 'standard commodity', then a Cadillac will have a consumption units price of 1000.

We could be even more direct about this. Money can itself be thought of as a good and, by convention, one unit of this money good has a price of one unit. The money good serves as a numeraire in terms of which the relative prices of all other goods are expressed. So one pair of Levi's has a consumption units price of 40, or a money price of \$40.

What is the conclusion of all this? Essentially, it is that prices can be thought of equally well in terms of consumption units or money units. They are alternative but equivalent ways of measuring some quantity. Throughout this book, the terms 'benefits' and 'costs' are usually measured in units of money or its consumption equivalent. But we sometimes refer to prices in utils – units of utility – rather than in money/consumption units. It is here that matters may be a little baffling. But this turns out to be a very simple notion. Economists make extensive use of the utility function:

$$U = U(C)$$

where U is units of utility and C is units of consumption. Now suppose that the utility function were of the simple linear form $U = kC$ where k is some constant, positive number. Then units of utility are simply a multiple of units of consumption. So if $k = 2$, three units of consumption are equivalent to six units of utility, and so on.

But the utility function may be non-linear. Indeed, it is often assumed that utility rises with consumption but at a decreasing rate. One form of utility function that satisfies this assumption is $U = \log(C + 1)$, with log denoting the common logarithmic operator, and in which the argument of the function includes the additive constant 1 to keep utility non-negative. The chart in Figure 14.2 shows the relationship between utility and consumption for this particular utility function.

It is equally valid to refer to prices in utility units as in any other units. From Figure 14.2 it is clear that a utility price of 2 corresponds to a 'consumption' (or money) price of approximately

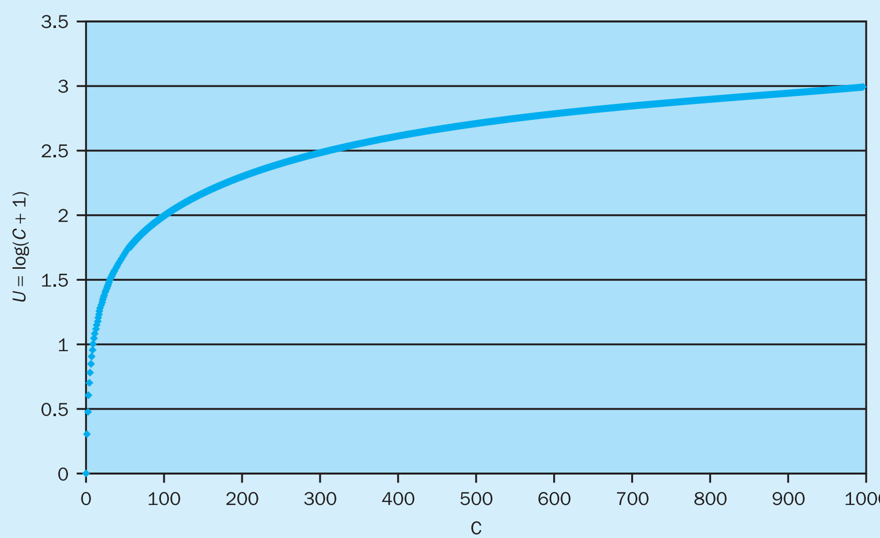


Figure 14.2 The logarithmic utility function

Box 14.2 *continued*

100 (in fact, 99 exactly). Also, a consumption price of 999 corresponds to a utility price of 3. What is the consumption units price equivalent to a price of 2.5 units of utility? (Use a calculator to find the answer, or read it off approximately from the diagram.)

Which units prices are measured in will depend on how the problem has been set up. In the chapters on resource depletion, what is being maximised is social welfare; given that the SWF is specified as a sum of utilities (of different people or different generations), it seems natural to denominate it in utility units as well, although our discussion makes it clear that we could convert units from utility into money/consumption terms if we wished to do so. In other parts of the book, what is being maximised is net benefit. That measure is typically constructed in consumption (or money income) units, and so it is natural to use money prices when dealing with problems set up in this way.

In conclusion, it is up to us to choose which units are most convenient. And provided we know what the utility function is (or are willing to make an assumption about what it is) then we can always move from one to the other as the occasion demands.

Now we are in a position to interpret the four conditions 14.14. First recall from the discussions in Chapters 4 and 11 that for any resource to be allocated efficiently, two kinds of conditions must be satisfied: static and dynamic efficiency. The first two of these conditions – 14.14a and 14.14b – are the static efficiency conditions that arise in this problem; the latter two are the dynamic efficiency conditions which must be satisfied. These are examined in a moment. The first two conditions – 14.14a and 14.14b – also implicitly define an optimal solution to this problem. We shall explain what this means shortly.

14.5.2 The static and dynamic efficiency conditions

You will recall from our discussions in Chapters 5 and 11 that the efficient allocation of any resource consists of two aspects.

14.5.2.1 The static efficiency conditions

As with any asset, static efficiency requires that, in each use to which a resource is put, the marginal value of the services from it should be equal to the marginal value of that resource stock *in situ*. This ensures that the marginal net benefit (or marginal value) to society of the resource should be the same in all its possible uses.

Inspection of equations 14.14a and 14.14b shows that this is what these equations do imply. Look first at equation 14.14a. This states that, in each period, the marginal utility of consumption $U_{C,t}$ must be equal to the shadow price of capital ω_t (remembering that prices are measured in units of utility here). A marginal unit of output can be used for consumption now (yielding $U_{C,t}$ units of utility) or added to the capital stock (yielding an amount of capital value ω_t in utility units). An efficient outcome will be one in which the marginal net benefit of using one unit of output for consumption is equal to its marginal net benefit when it is added to the capital stock.

Equation 14.14b states that the value of the marginal product of the natural resource must be equal to the marginal value (or shadow price) of the natural resource stock. This shadow price is, of course, P_t. The value of the marginal product of the resource is the marginal product in units of output (i.e. $Q_{R,t}$) multiplied by the value of one unit of output, ω_t. But we have defined ω_t as the price of a unit of capital; so why is this the value of one unit of output? The reason is simple. In this economy, units of output and units of capital are in effect identical (along an optimal path). Any output that is not consumed is added to capital. So we can call ω_t either the value of a marginal unit of capital or the value of a marginal unit of output.

14.5.2.2 The dynamic efficiency conditions

Dynamic efficiency requires that each asset or resource earns the same rate of return, and that this rate of return is the same at all points in time, being equal to the social rate of discount. Equations 14.14c and 14.14d ensure that dynamic efficiency is satisfied. Consider first equation 14.14c. Dividing each side by P we obtain $\dot{P}_t/P_t = \rho$ which states that the growth rate of the shadow price of the natural resource (that is, its own rate of return) should equal the social utility discount rate. Finally, dividing both sides of 14.14d by ω, we obtain

$$\frac{\dot{\omega}_t}{\omega_t} + Q_{K,t} = \rho$$

which states that the return to physical capital (its capital appreciation plus its marginal productivity) must equal the social discount rate.

14.5.3 Hotelling's rule: two interpretations

Equation 14.14c is known as Hotelling's rule for the extraction of non-renewable resources. It is often expressed in the form

$$\frac{\dot{P}_t}{P_t} = \rho \tag{14.15}$$

The Hotelling rule is an intertemporal efficiency condition which must be satisfied by any efficient process of resource extraction. In one form or another, we shall return to the Hotelling rule during this and the following three chapters. One interpretation of this condition was offered above. A second can be given in the following way. First rewrite equation 14.15 in the form used earlier, that is

$$\dot{P}_t = \rho P_t \tag{14.16}$$

By integration of equation 14.16 we obtain[6]

$$P_t = P_0 e^{\rho t} \tag{14.17}$$

P_t is the undiscounted price of the natural resource. The discounted price is obtained by discounting P_t at the social utility discount rate ρ. Denoting the discounted resource price by P^*, we have

$$P_t^* = P_t e^{-\rho t} = P_0 \tag{14.18}$$

Equation 14.18 states that the discounted price of the natural resource is constant along an efficient resource extraction path. In other words, Hotelling's rule states that the discounted unit value of the resource should be the same at all dates. But this is merely a special case of a general asset–efficiency condition; the discounted (or present value) price of any efficiently managed asset will remain constant over time. This way of interpreting Hotelling's rule shows that there is nothing special about natural resources *per se* when it comes to thinking about efficiency. A natural resource is an asset. All efficiently managed assets will satisfy the condition that their discounted prices should be equal at all points in time. If we had wished to do so, the Hotelling rule could have been obtained directly from this general condition.

Before moving on, note the effect of changes in the social discount rate on the optimal path of resource price. The higher is ρ, the faster should be the rate of growth of the natural resource price. This result is eminently reasonable given the two interpretations we have offered of the Hotelling rule. Its implications will be explored in the following chapter.

14.5.4 The growth rate of consumption

We show in Appendix 14.2 that equations 14.14a and 14.14d can be combined to give

$$\frac{\dot{C}}{C} = \frac{Q_K - \rho}{\eta}$$

The term η, the elasticity of marginal utility with respect to consumption, is necessarily positive under the assumptions we have made about the utility function. It therefore follows that

$$\frac{\dot{C}}{C} > 0 \Leftrightarrow Q_K > \rho$$

$$\frac{\dot{C}}{C} = 0 \Leftrightarrow Q_K = \rho$$

$$\frac{\dot{C}}{C} < 0 \Leftrightarrow Q_K < \rho$$

Some intuition may help to understand these relations. The social discount rate, ρ, reflects impatience for future consumption; Q_K (the marginal product of capital) is the pay-off to delayed consumption. The relations imply that along an optimal path:

[6] A reader of an earlier edition, Alexandre Croutzet, pointed out that obtaining equation 14.17 from equation 14.16 can probably be done by integration but it is not straightforward. He noted that it may be helpful to recall that 14.16 is a first-order linear homogenous differential equation and that 14.17 is its definite solution using $P(t = 0) = P_o$.

(a) consumption is increasing when 'pay-off' is greater than 'impatience';
(b) consumption is constant when 'pay-off' is equal to 'impatience';
(c) consumption is decreasing when 'pay-off' is less than 'impatience'.

Therefore, consumption is growing over time along an optimal path if the marginal product of capital (Q_K) exceeds the social discount rate (ρ); consumption is constant if $Q_K = \rho$; and consumption growth is negative if the marginal product of capital is less than the social discount rate. This makes sense, given that:

(a) when 'pay-off' is greater than 'impatience', the economy will be accumulating K and hence growing;
(b) when 'pay-off' and 'impatience' are equal, K will be constant;
(c) when 'pay-off' is less than 'impatience', the economy will be running down K.

14.5.5 Optimality in resource extraction

The astute reader will have noticed that we have described the Hotelling rule (and the other conditions described above) as an efficiency condition. But a rule that requires the growth rate of the price of a resource to be equal to the social discount rate does not give rise to a unique price path. This should be evident by inspection of Figure 14.3, in which two different initial prices, say 1 util and 2 utils, grow over time at the same discount rate, say 5%. If ρ were equal to 5%, each of these paths – and indeed an infinite quantity of other such price paths – satisfies Hotelling's rule, and so they are all efficient paths. But only one of these price paths can be optimal, and so the Hotelling rule is a necessary but not a sufficient condition for optimality.

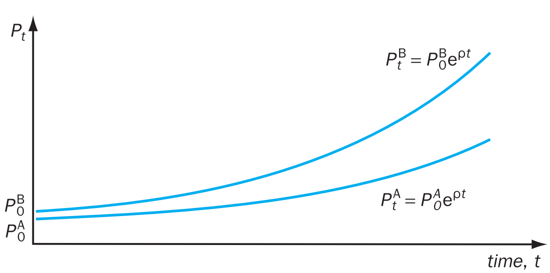


Figure 14.3 Two price paths, each satisfying Hotelling's rule

How do we find out which of all the possible efficient price paths is the optimal one? An optimal solution requires that all of the conditions listed in equations 14.14a–d, together with initial conditions relating to the stocks of capital and resources and terminal conditions (how much stocks, if any, should be left at the terminal time), are satisfied simultaneously. So Hotelling's rule – one of these conditions – is a necessary but not sufficient condition for an optimal allocation of natural resources over time.

Let us think a little more about the initial and final conditions for the natural resource that must be satisfied. There will be some initial resource stock; similarly, we know that the resource stock must converge to zero as elapsed time passes and the economy approaches the end of its planning horizon. If the initial price were 'too low', then this would lead to 'too large' amounts of resource use in each period, and all the resource stock would become depleted in finite time (that is, before the end of the planning horizon). Conversely, if the initial price were 'too high', then this would lead to 'too small' amounts of resource use in each period, and some of the resource stock would (wastefully) remain undepleted at the end of the planning horizon. This suggests that there is one optimal initial price that would bring about a path of demands that is consistent with the resource stock just becoming fully depleted at the end of the planning period.

In conclusion, we can say that while equations 14.14 are each efficiency conditions, taken jointly as a set (together with initial values for K and S) they implicitly define an optimal solution to the optimisation problem, by yielding unique time paths for K_t and R_t and their associated prices that maximise the social welfare function.

PART II EXTENDING THE MODEL TO INCORPORATE EXTRACTION COSTS AND RENEWABLE RESOURCES

In our analysis of the depletion of resources up to this point, we have ignored extraction costs. Usually the extraction of a natural resource will be costly. So it is desirable to generalise our analysis to allow for the presence of these costs.

It seems likely that total extraction costs will rise as more of the material is extracted. So, denoting total extraction costs as Γ and the amount of the resource being extracted as R, we would expect that Γ will be a function of R. A second influence may also affect extraction costs. In many circumstances, costs will depend on the size of the remaining stock of the resource, typically rising as the stock becomes more depleted. Letting S_t denote the size of the resource stock at time t (the amount remaining after all previous extraction) we can write extraction costs as

$$\Gamma_t = \Gamma(R_t, S_t) \tag{14.19}$$

To help understand what the presence of the stock term in equation 14.19 implies about extraction costs, look at Figure 14.4. This shows three possible relationships between total extraction costs and the remaining resource stock size for a constant level of resource extraction. The relationship denoted (i) corresponds to the case where the total extraction cost is independent of the stock size. In this case, the extraction cost function collapses to the simpler form $\Gamma_t = \Gamma_1(R_t)$ in which extraction costs depend only on the quantity extracted per period of time. In case (ii), the costs of extracting a given quantity of the resource increase linearly as the stock becomes increasingly depleted. $\Gamma_S = \partial\Gamma/\partial S$ is then a constant negative number. Finally, case (iii) shows the costs of extracting a given quantity of the resource increasing at an increasing rate as S falls towards zero; Γ_S is negative but not constant, becoming larger in absolute value as the resource stock size falls. This third case is the most likely one for typical non-renewable resources. Consider, for example, the cost of extracting oil. As the available stock more closely approaches zero, capital equipment is directed to exploiting smaller fields, often located in geographically difficult land or marine areas. The quality of resource stocks may also fall in this process, with the best fields having been exploited first. These and other similar reasons imply that the cost of extracting an additional barrel of oil will tend to rise as the remaining stock gets closer to exhaustion.

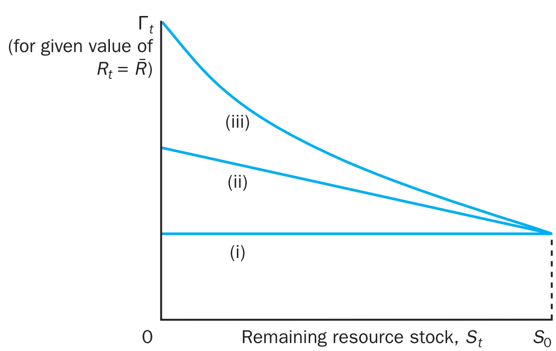


Figure 14.4 Three possible examples of the relationship between extraction costs and remaining stock for a fixed level of resource extraction, R

14.6 The optimal solution to the resource depletion model incorporating extraction costs

The problem we now wish to solve can be expressed as follows:

Select values for the choice variables C_t and R_t for $t = 0, \ldots, \infty$ so as to maximise

$$W = \int_{t=0}^{t=\infty} U(C_t)e^{-\rho t}dt$$

subject to the constraints

$$\dot{S}_t = -R_t$$

and

$$\dot{K}_t = Q(K_t, R_t) - C_t - \Gamma(R_t, S_t)$$

Comparing this with the description of the optimisation problem for the simple model, one difference can be found. The differential equation for $\dot{K}$ now includes extraction costs as an additional term. Output produced through the production function $Q(K, R)$ should now be thought of as gross output. Extraction costs make a claim on this gross output, and net-of-extraction cost output is gross output minus extraction costs (that is, $Q - \Gamma$).

The solution to this problem is once again obtained using the maximum principle of optimal control. If you wish to go through the derivations, you should follow the steps in Appendix 14.2, but this time ensuring that you take account of the extraction cost term which will now appear in the differential equation for $\dot{K}$, and so also in the Hamiltonian. From this point onwards, we shall omit time subscripts for simplicity of notation unless their use is necessary in a particular context. The necessary conditions for a social welfare optimum now become:

$$U_C = \omega \tag{14.20a}$$

$$P = \omega Q_R - \omega\Gamma_R \tag{14.20b}$$

$$\dot{P} = \rho P + \omega\Gamma_S \tag{14.20c}$$

$$\dot{\omega} = \rho\omega - Q_K\omega \tag{14.20d}$$

Note that two of these four equations – 14.20a and 14.20d – are identical to their counterparts in the solution to the simple model we obtained earlier, and the interpretations offered then need not be repeated. However, the equations for the resource net price and for the rate of change of net price differ. Some additional comment is warranted on these two equations.

First, with extraction costs now entering the analysis, it is necessary to distinguish between two kinds of price: gross price and net price. This distinction follows from that we have just made between gross and net output. These two measures of the resource price are related by net price being equal to gross price less marginal cost. Equation 14.20b can be seen in this light:

P_t	$= \omega_t Q_R$	less $\omega_t \Gamma_R$
Net price	= Gross price	less Marginal cost

The term $\omega_t\Gamma_R$ is the value of the marginal extraction cost, being the product of the impact on output of a marginal change in resource extraction and the price of capital (which, as we saw earlier, is also the value of one unit of output). Equation 14.20b can be interpreted in a similar way to that given for equation 14.14b. That is, the value of the marginal net product of the natural resource ($\omega Q_R - \omega\Gamma_R$, the marginal gross product less the marginal extraction cost) must be equal to the marginal value (or shadow net price) of a unit of the natural resource stock, P.

If profit-maximising firms in a competitive economy were extracting resources, these marginal costs would be internal to the firm and the market price would be identical to the gross price. Note that the level of the net (and the gross) price is only affected by the effect of the extraction rate, R, on costs. The stock effect does not enter equation 14.20b.

The stock effect on costs does, however, enter equation 14.20c for the rate of change of the net price of the resource. This expression is the Hotelling rule, but now generalised to take account of extraction costs. The modified Hotelling rule (equation 14.20c) is:

$$\dot{P} = \rho P + \Gamma_S\omega$$

Given that $\Gamma_S = \partial\Gamma/\partial S$ is negative (resource extraction is more costly the smaller is the remaining stock), efficient extraction over time implies that the rate of increase of the resource net price should be lower where extraction costs depend upon the resource stock size. A little reflection shows that this is eminently reasonable. Once again, we work with an interpretation given earlier. Dividing equation 14.20c by the resource net price we obtain

$$\rho = \frac{\dot{P}}{P} - \frac{\Gamma_S\omega}{P}$$

which says that, along an efficient price path, the social rate of discount should equal the rate of return from holding the resource (which is given by its rate of capital appreciation, plus the present value of the extraction cost increase that is avoided by not extracting an additional unit of the stock, $-\Gamma_S\omega/P$.)

There is yet another possible interpretation of equation 14.20c. To obtain this, first rearrange the equation to the form:

$$\rho P = \dot{P} - \Gamma_S\omega \tag{14.20*}$$

The left-hand side of 14.20* is the marginal cost of not extracting an additional unit of the resource; the right-hand side is the marginal benefit from not extracting an additional unit of the resource. At an efficient (and at an optimal) rate of resource use, the marginal costs and benefits of resource use are balanced at each point in time. How is this interpretation obtained? Look first at the left-hand side. The net price of the resource, P, is the value that would be obtained by the resource owner were he or she to extract and sell the resource in the current period. With ρ being the social utility discount rate, ρP is the utility return forgone by not currently extracting one unit of the resource, but deferring that extraction for one period. This is sometimes known as the holding cost of the resource stock. The right-hand side contains two components. $\dot{P}$ is the capital appreciation of one unit of the unextracted resource; the second component, $-\Gamma_S\omega$, is a return in the form of a postponement of a cost increase that would have occurred if the additional unit of the resource had been extracted.

Finally, note that whereas the static efficiency condition 14.20b is only affected by the current extraction rate, R, the dynamic efficiency condition (Hotelling's rule, 14.20c) is only affected by the stock effect on costs.

In conclusion, the presence of costs related to the level of resource extraction raises the gross price of the resource above its net price but has no effect on the growth rate of the resource net price. Note that net price is what we referred to as rent in Chapter 3: it is also sometimes referred to as royalty. In contrast, a resource stock size effect on extraction costs will slow down the rate of growth of the resource net price. In most circumstances, this implies that the resource net price has to be higher initially (but lower ultimately) than it would have been in the absence of this stock effect. As a result of higher initial prices, the rate of extraction will be slowed down in the early part of the time horizon, and a greater quantity of the resource stock will be conserved (to be extracted later).

14.7 Generalisation to renewable resources

We reserve a lengthy analysis of the allocation of renewable resources until Chapters 17 and 18, but it will be useful at this point to suggest the way in which the analysis can be undertaken. To do so, first note that we have been using S to represent a fixed and finite stock of a non-renewable resource. The total use of the resource over time was constrained to be equal to the fixed initial stock. This relationship arises because the natural growth of non-renewable resources is zero except over geological periods of time. Thus we wrote

$$S_t = S_0 - \int_{\tau=0}^{\tau=t} R_\tau \mathrm{d}\tau \Rightarrow \dot{S}_t = -R_t$$

However, the natural growth of renewable resources is, in general, non-zero. A simple way of modelling this growth is to assume that the amount of growth of the resource, G_t, is some function of the current stock level, so that $G_t = G(S_t)$.[7] Given this we can write the relationship between the change in the resource stock and the rate of extraction (or harvesting) of the resource as

$$\dot{S}_t = G(S_t) - R_t \tag{14.21}$$

Not surprisingly, the efficiency conditions required by an optimal allocation of resources are now different from the case of non-renewable resources. However, a modified version of the Hotelling rule for rate of change of the net price of the resource still applies, given by

$$\dot{P} = \rho P - PG_s \tag{14.22}$$

where $G_S = \mathrm{d}G/\mathrm{d}S$, and in which we have assumed, for simplicity, that harvesting does not incur costs, nor that any natural damage results from the harvesting and use of the resource. Inspection of the modified Hotelling rule for renewable resources (equation 14.22) demonstrates that the rate at which the net price should change over time depends upon G_S, the rate of change of resource growth with respect to changes in the stock of the resource. We will not attempt to interpret equation 14.22 here as that is best left until we examine renewable resources in detail later. However, it is worth saying a few words about steady-state resource harvesting.

A steady-state harvesting of a renewable resource exists when all stocks and flows are constant over time. In particular, a steady-state harvest will be one in which the harvest level is fixed over time and is equal to the natural amount of growth of the resource stock. Additions to and subtractions from the resource stock are thus equal, and the stock remains at a constant level over time. Now if the demand for the resource is constant over time, the resource net price will remain constant in a steady

[7] Note that we are using G to refer to a growth rate itself, rather than to the level of a variable; given this, we do not need a dot above G.

state, as the quantity harvested each period is constant. Therefore, in a steady state, $\dot{P} = \rho P - PG_s = 0$. So in a steady state, the Hotelling rule simplifies to

$$\rho P = PG_S \tag{14.23}$$

and so

$$\rho = G_S \tag{14.24}$$

It is common to assume that the relationship between the resource stock size, S, and the growth of the resource, G, is as indicated in Figure 14.5. This relationship is explained more fully in Chapter 17. As the stock size increases from zero, the amount of growth of the resource rises, reaches a maximum, known as the maximum sustainable yield (MSY), and then falls. Note that $G_S = \mathrm{d}G/\mathrm{d}S$ is the slope at any point of the growth–stock function in Figure 14.5.

We can deduce that if the social utility discount rate ρ were equal to zero, then the efficiency condition of equation 14.24 could only be satisfied if the steady-state stock level is $\hat{S}$, and the harvest is the MSY harvest level. On the other hand, if the social discount rate was positive (as will usually be the case), then the efficiency condition requires that the steady-state stock level be less than $\hat{S}$. At the stock level S^*, for example, G_S is positive, and would be an efficient stock level, yielding a sustainable yield of G^*, if the discount rate were equal to this value of G_S. Full details of the derivation of this and other results relating to the Hotelling rule are given in Appendix 14.3.

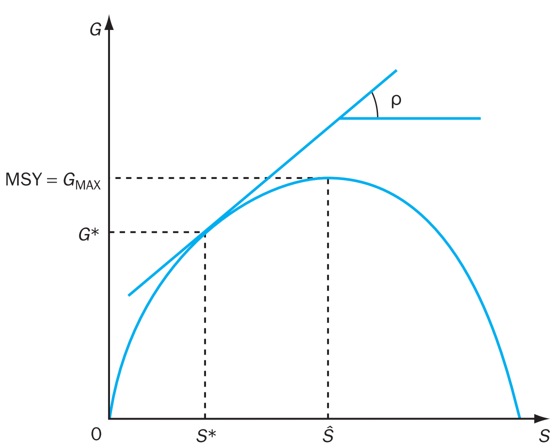


Figure 14.5 The relationship between the resource stock size, *S*, and the growth of the resource, *G*

14.8 Complications

The model with which we began in this chapter was one in which there was a single, known, finite stock of a non-renewable resource. Furthermore, the whole stock was assumed to have been homogeneous in quality. In practice, both of these assumptions are often false. Rather than there existing a single non-renewable resource, there are many different classes or varieties of non-renewable resource, some of which may be relatively close substitutes for others (such as oil and natural gas, and iron and aluminium). While it may be correct to assume that there exists a given finite stock of each of these resource stocks in a physical sense, the following situations are likely:

1. The total stock is not known with certainty.
2. New discoveries increase the known stock of the resource.
3. A distinction needs to be drawn between the physical quantity of the stock and the economically viable stock size.
4. Research and development, and technical progress, take place, which can change extraction costs, the size of the known resource stock, the magnitude of economically viable resource deposits, and estimates of the damages arising from natural resource use.

Furthermore, even when we focus on a particular kind of non-renewable resource, the stock is likely to be heterogeneous. Different parts of the total stock are likely to be uneven in quality, or to be located in such a way that extraction costs differ for different portions of the stock.

By treating all non-renewable resources as one composite good, our analysis in this chapter had no need to consider substitutes for the resource in question (except, of course, substitutes in the form of capital and labour). But once our analysis enters the more complex world in which there are a variety of different non-renewable resources which are substitutable for one another to some degree, analysis inevitably becomes more complicated. One particular issue of great potential importance is the presence of backstop technologies (see Chapter 15). Suppose that we are currently using some non-renewable resource for a particular purpose – perhaps for energy production. It may well be the case that another resource exists that can substitute entirely for the resource we are considering, but may not be used at present because its cost is relatively high. Such a resource is known as a backstop technology. For example, renewable power sources such as wind energy are backstop alternatives to fossil-fuel-based energy.

The existence of a backstop technology will set an upper limit on the level to which the price of a resource can go. If the cost of the 'original' resource were to exceed the backstop cost, users would switch to the backstop. So even though renewable power is probably not currently economically viable, at least not on a large scale, it would become so at some fossil-fuel cost, and so the existence of a backstop will lead to a price ceiling for the original resource.

Each of the issues we have raised in this section, and which we have collectively called 'complications', need to be taken account of in any comprehensive account of resource use. We shall do so in the next four chapters.

14.9 A numerical application: oil extraction and global optimal consumption

In this section we present a simple, hypothetical numerical application of the theory developed above. You may find the mathematics of the solution given in Box 14.3 a little tedious; if you wish to avoid the maths, just skip the box and proceed to Table 14.1 and Figures 14.6–14.8 at the end of the section where the results are laid out. (The derivation actually uses the technique of dynamic optimisation explained in Appendix 14.1, but applied in this case to a discrete-time model.)

Box 14.3 Solution of the dynamic optimisation problem using the maximum principle

The current value of the Hamiltonian is

$$H_t = U(C_t) + P_{t+1}(-R_t) + \omega_{t+1}(Q(K_t, R_t) - C_t)$$
$$(t = 0, 1, \ldots, T-1)$$

where P_t is the shadow price of oil (at time t), and ω_t is the shadow price of capital. The four necessary conditions for an optimum are:

1. $P_{t+1} - P_t = \rho P_1 - \dfrac{\partial H_t}{\partial S_t} = \rho P_t$

which implies Hotelling's efficiency rule

$$P_{t+1} = (1+\rho)P_t \quad (t = 1, \ldots, T)$$

2. $\omega_{t+1} - \omega_t = \rho\omega_t - \dfrac{\partial H_t}{\partial K_t} = \rho\omega_t - \omega_{t+1}Q_{Kt}$

which implies

$$\omega_{t+1} = \frac{1+\rho}{1+Q_{Kt}}\omega_t \quad (t = 1. \ldots, T)$$

where $Q_{Kt} = Q_{Kt}(K_t, R_t) = \dfrac{\partial Q(K_t, R_t)}{\partial Kt}$

3. $\dfrac{\partial H_t}{\partial R_t} = 0 = -P_{t+1} + \omega_{t+1}Q_{Rt} \quad (t = 0, 1 \ldots, T-1)$

where $Q_{Rt} = Q_{Rt}(K_t, R_t) = \dfrac{\partial Q(K_t, R_t)}{\partial R_t}$

4. $\dfrac{\partial H_t}{\partial C_t} = 0 = U'(C_t) - \omega_{t+1} \quad (t = 0, 1 \ldots, T-1)$

where $U'(C_t) = \dfrac{\mathrm{d}U(C_t)}{\mathrm{d}C_t}$

Since we know $S_T = K_T = 0$, there are $6T$ unknowns in this problem as given below. The unknowns are:

Number	$T+1$	$T+1$	$T-1$	$T-1$
Unknowns	$\{P_t\}_1^{T+1}$	$\{\omega_t\}_1^{T+1}$	$\{K_t\}_1^{T-1}$	$\{S_t\}_1^{T-1}$
	T	T	$= 6T$	
	$\{C_t\}_0^{T-1}$	$\{R_t\}_0^{T-1}$		

We have $6T$ equations to solve for these $6T$ unknowns. The equations are:

$U'(C_t) = \omega_{t+1} \quad (t = 0, 1, \ldots, T-1)$ [T equations]

$P_{t+1} = \omega_{t+1}Q_{Rt} \quad (t = 0, 1, \ldots, T-1)$ [T equations]

$S_{t+1} = S_t - R_t \quad (t = 0, 1, \ldots, T-1)$ [T equations]

$P_{t+1} = (1+\rho)P_t \quad (t = 1, \ldots, T)$ [T equations]

$\omega_{t+1} = \dfrac{1+\rho}{1+Q_{Kt}}\omega_t \quad (t = 1. \ldots, T)$ [T equations]

$K_{t+1} = K_t + Q(K_t, R_t) - C_t \quad (t = 0, 1 \ldots, T-1)$ [T equations]

Table 14.1 Numerical solution to the oil extraction and optimal consumption problem

Welfare (p.v.) = 46.67668

Time	C_t	$Q(K_t, R_t)$	K_t	R_t	S_t	$\omega(t{+}+1)$	$P(t+1)$	$\partial Q/\partial K(t)$
1990s	3.7342	18.0518	4.9130	2.2770	11.5000	0.2678	0.2123	3.3069
2000s	13.6819	60.8347	19.2306	1.9947	9.2230	0.0731	0.2229	2.8471
2010s	45.3301	182.8353	66.3834	1.7232	7.2283	0.0221	0.2341	2.4788
2020s	137.2777	493.8224	203.8886	1.4637	5.5051	0.0073	0.2458	2.1798
2030s	383.6060	1204.3708	560.4334	1.2167	4.0413	0.0026	0.2581	1.9341
2040s	997.3350	2654.7992	1381.1982	0.9824	2.8247	0.0010	0.2710	1.7299
2050s	2430.1080	5261.7198	3038.6624	0.7611	1.8423	0.0004	0.2845	1.5584
2060s	5584.8970	9217.1125	5870.2742	0.5525	1.0812	0.0002	0.2987	1.4131
2070s	12174.5621	13608.6578	9502.4897	0.3564	0.5287	0.0001	0.3137	1.2889
2080s	25298.4825	14361.8971	10936.5854	0.1724	0.1724	0.0000	0.3293	1.1819
2090s			0.0	0.0000	0.0000	0.0000	0.3548	
	× 10 trillion US$			× 100 billion barrels				
% Growth rates:								
2000	266.3899	237.0006	291.4222	−12.4012	−19.8004	−74.0064	5.0000	
2010	231.3150	200.5443	245.1973	−13.6071	−21.6271	−71.2545	5.0000	
2020	202.8399	170.0914	207.1378	−15.0607	−23.8402	−68.5517	5.0000	
2030	179.4380	143.8874	174.8723	−16.8783	−26.5885	−65.9180	5.0000	
2040	159.9894	120.4304	146.4518	−19.2530	−30.1053	−63.3685	5.0000	
2050	143.6602	98.1965	120.0019	−22.5320	−34.7797	−60.9136	5.0000	
2060	129.8209	75.1730	93.1861	−27.4081	−41.3110	−58.5599	5.0000	
2070	117.9908	47.6456	61.8747	−35.4951	−51.0972	−56.3110	5.0000	
2080	107.7979	5.5350	15.0918	−51.3311	−67.3996	−54.1679	5.0000	

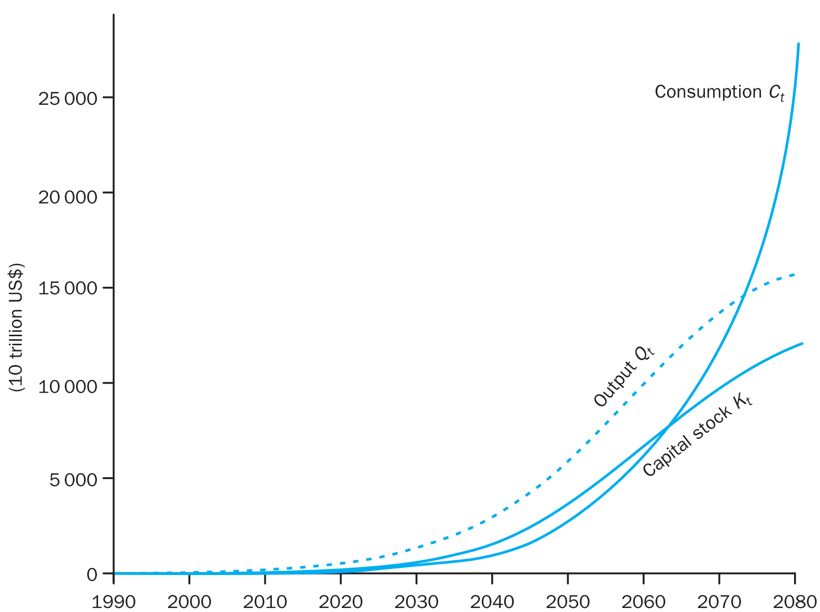


Figure 14.6 Numerical application: optimal time paths of output, consumption and capital stock

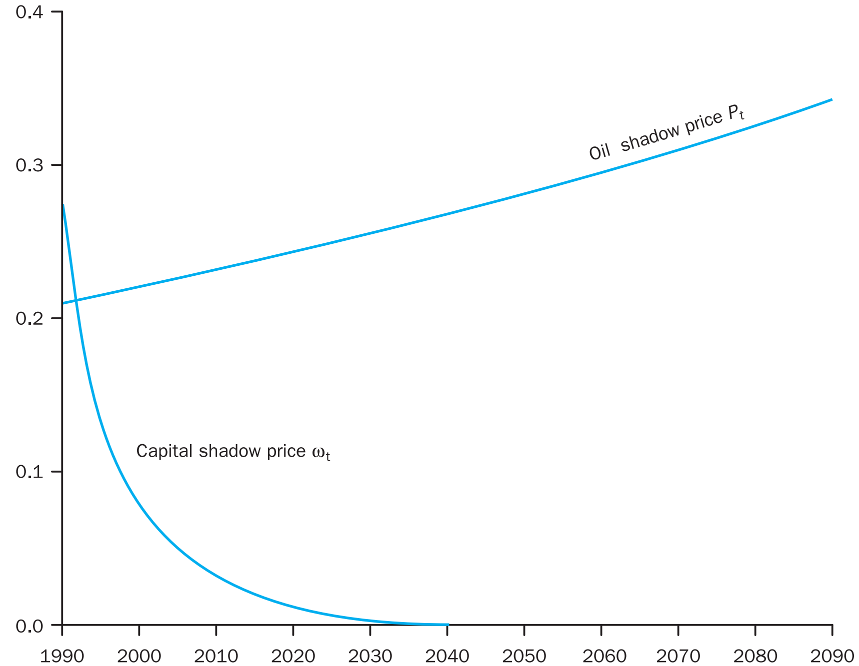


Figure 14.7 Numerical application: optimal time paths of the oil and capital shadow prices

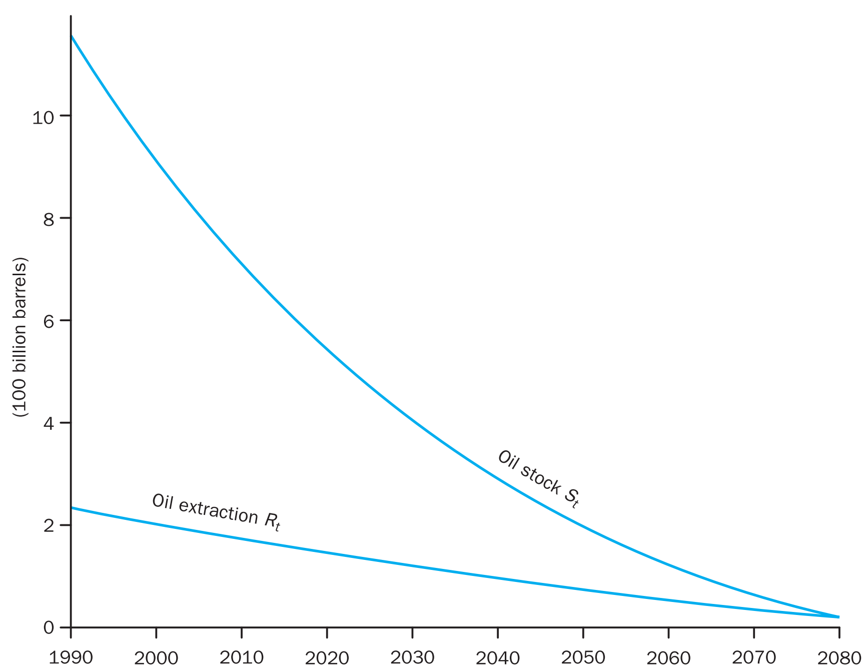


Figure 14.8 Numerical application: optimal time paths of oil extraction and the remaining oil stock

Suppose that the welfare function to be maximised is

$$W = \sum_{t=0}^{t=T-1} \frac{U(C_t)}{(1+\rho)^t}$$

where C_t is the global consumption of goods and services at time t; U is the utility function, with $U(C_t) = \log C_t$; ρ is the utility discount rate; and T is the terminal point of the optimisation period. The relevant constraints are

$$S_{t+1} = S_t - R_t$$

$$K_{t+1} = K_t + Q(K_t R_t) - C_t$$

$$S_T = K_T = 0$$

S_0 and K_0 are given.

S_t denotes the stock of oil; R_t is the rate of oil extraction; K_t is the capital stock; and $Q(K_t, R_t) = AK_t^{0.9}R_t^{0.1}$ is a Cobb–Douglas production function, with A being a fixed 'efficiency' parameter. In this application, we assume that oil extraction costs are zero, and that there is no depreciation of the capital stock. Note that we assume that there are fixed initial stocks of the state variables (non-renewable resource and the capital stock), and that we specify that the state variables are equal to zero at the end of the optimisation period.

We also assume that a backstop technology exists that will replace oil as a productive input at the end (terminal point) of the optimisation period, $t = T$. This explains why we set $S_T = 0$, as there is no point having any stocks of oil remaining once the backstop technology has replaced oil in production. We assume that the capital stock, K_t, associated with the oil input, will be useless for the backstop technology, and therefore will be consumed completely by the end of the optimisation period, so $K_T = 0$.

Implicitly in this simulation, we assume that a new capital stock, appropriate for the backstop technology, will be accumulated out of the resources available for consumption. So C_t in this model should be interpreted as consumption plus, where necessary, new additions to the (backstop) capital stock. The question of how much should be saved to accumulate this new capital stock is beyond the scope of our simple model.

As the notation will have made clear, this is a discrete-time model. We choose each period to be 10 years, and consider a 10-period ($t = 0, 1, \ldots, 9$) time horizon of 100 years beginning in 1990 ($t = 0$). The following data are used in the simulation:

Estimated world oil reserve = 11.5 (units of 100 billion barrels)
World capital stock = 4.913 (units of 10 trillion \$US)
Efficiency parameter $A = 3.968$
Utility discount rate = 5%

The value of the efficiency parameter is estimated under the assumption that world aggregate output over the 1980s was \$US179.3 trillion, and aggregate oil extraction was 212.7 billion barrels.

We cannot obtain an analytical solution for this problem. But it is possible to solve the problem numerically on a computer. Table 14.1 and Figures 14.6–14.8 show the numerical solution. Figure 14.6 shows that consumption rises exponentially through the whole of the optimisation period (the figure only shows the first part of that period, from 1990 up to 2080); output (Q_t) also rises continuously, but its growth rate slows down towards the end of the period shown in the figure; this arises because at the terminal point (not shown in these figures), the capital stock has to fall to zero for the reason indicated above.

In Figure 14.7 we observe the shadow price growing over time at the rate ρ, and so satisfying Hotelling's rule; the shadow price of capital, ω, falls continuously through time. In Figure 14.8, we see that the oil stock falls gradually from its initial level towards zero; note that as the shadow price of oil rises over time, so the rate of extraction falls towards zero. Not surprisingly, the optimal solution will require that the rate of extraction goes to zero at exactly the same point in time, call it t_f, that the stock is completely exhausted. Note that within 100 years, the oil stock has fallen to a level not significantly different from zero; it is optimal to deplete the stock of oil fairly rapidly in this model. What happens after the year 2090? You should now try to deduce the answer to this question.

Summary

- In this chapter, we have constructed a simple economic model of optimal resource depletion, and studied several of its variants. The framework we use is a version of the so-called optimal growth models, built around a production function in which natural resources are inputs into the production process.
- The solution to an optimal resource depletion model should be one in which a set of static and intertemporal efficiency conditions – as discussed in Chapters 5 and 11 – are satisfied. In addition, optimality requires that, of the many possible efficient depletion paths that may be available, the one chosen is that which maximises the relevant objective function (in this case the intertemporal social welfare function).
- The characteristics of a socially optimal pattern of natural resource use over time will depend on the particular nature of the social welfare function that is deemed to be appropriate. In this chapter – as generally throughout the text – the special case chosen is that of the utilitarian social welfare function, in which future utility is discounted at the positive rate ρ.
- Given that the objective function used here is specified in terms of units of utility, all prices and values referred to in this chapter are also specified in units of utility. This poses a practical problem in that utility is an unobservable variable. However, given the form of the utility function $U = U(C)$, all results could be stated equivalently in terms of units of consumption (or income).
- One efficiency property to which resource economists pay considerable attention is the so-called Hotelling rule. This intertemporal efficiency condition requires that the real rate of return to a resource owner should equal the social discount rate.
- For a non-renewable resource available in known, fixed quantity, the Hotelling rule implies that the net price of the resource, specified in utility units, should grow at the proportionate rate ρ.
- The Hotelling rule has general applicability and does not apply only to non-renewable resources. We briefly state how the rule applies to renewable resources. This will be examined at length in Chapter 17. Although we did not discuss it in this chapter, the rule also applies where resource extraction or use generates adverse external effects. This case is examined at length in Chapter 16.

Further reading

The mathematics underlying our analyses is presented simply and clearly in Chiang (1984, 1992). Kamien and Schwartz (1991) is also an excellent reference for optimal control theory. Excellent advanced-level presentations of the theory of efficient and optimal resource depletion can be found in Baumol and Oates (1988), Dasgupta and Heal (1979), and Heal (1981). Kolstad and Krautkraemer (1993) is particularly insightful and relatively straightforward. Dasgupta and Heal (1974) is also a comprehensive study, and is contained along with several other useful (but difficult) papers in the May 1974 special issue of the *Review of Economic Studies*.

Less difficult presentations are given in Hartwick and Olewiler (1998), Anderson (1985, 1991), and the Fisher and Peterson survey article in the March 1976 edition of the *Journal of Economic Literature*. For an application of this theory to the greenhouse effect, see Barbier (1989b) or Nordhaus (1982, 1991a). Barbier (1989a) provides a critique of the conventional theory of resource depletion, and other critical discussions are found in Common (1995) and Common and Perrings (1992).

Discussion questions

1. Are non-renewable resources becoming more or less substitutable by other productive inputs with the passage of time? What are the possible implications for efficient resource use of the elasticity of substitution between non-renewable resources and other inputs becoming
   (a) higher, and
   (b) lower
   with the passage of time?
2. Discuss the possible effects of technical progress on resource substitutability.
3. Recycling of non-renewable resources can relax the constraints imposed by finiteness of non-renewable resources. What determines the efficient amount of recycling for any particular economy?

Problems

1. Using the relationship

$$r = \rho + \eta\frac{\dot{C}}{C}$$

   demonstrate that if the utility function is of the special form $U(C) = C$, the consumption rate of discount (r) and the utility rate of discount are identical.
2. Using equation 14.15 in the text (that is, the Hotelling efficiency condition), demonstrate the consequences for the efficient extraction of a non-renewable resource of an increase in the social discount rate, ρ.
3. The simplest model of optimal resource depletion is the so-called 'cake-eating' problem in which welfare is a discounted integral of utility, utility is a function of consumption, and consumption is equal to the amount of the (non-renewable) resource extracted. That is:

$$W = \int_{t=0}^{t=\infty} U(C_t)\mathrm{e}^{-\rho t}\mathrm{d}t$$

$C_t = R_t$ and

$\dot{S}_t = -R_t$

   (a) Obtain the Hamiltonian, and the necessary first-order conditions for a welfare maximum.
   (b) Interpret the first-order conditions.
   (c) What happens to consumption along the optimal path?
   (d) What is the effect of an increase in the discount rate?

CHAPTER 15 The theory of optimal resource extraction: non-renewable resources

Behold, I have played the fool, and have erred exceedingly.

1 Samuel 26:21

Learning objectives

After the end of this chapter the reader should be able to

- understand the concept of non-renewable resources
- appreciate the distinctions between alternative measures of resource stock, such as base resource, resource potential and resource reserves
- understand the role of resource substitution possibilities and the ideas of a backstop technology and a resource choke price
- construct and solve simple discrete time and continuous time models of optimal resource depletion
- understand the meaning of a socially optimal depletion programme, and why this may differ from privately optimal programmes
- carry out simple comparative dynamic analysis in the context of resource depletion models, and thereby determine the consequences of changes in interest rates, known stock size, demand, price of backstop technology, and resource extraction costs
- compare resource depletion outcomes in competitive and monopolistic markets
- identify the consequences of taxes and subsidies on resource net prices and resource revenues
- understand the concept of natural resource scarcity, and be aware of a variety of possible measures of scarcity

Introduction

Non-renewable resources include fossil-fuel energy supplies – oil, gas and coal – and minerals – copper and nickel, for example. They are formed by geological processes over millions of years and so, in effect, exist as fixed stocks which, once extracted, cannot be renewed. One question is of central importance: what is the optimal extraction path over time for any particular non-renewable resource stock?

We began to answer this question in Chapter 14. There the optimal extraction problem was solved for a special case in which there was one homogeneous (uniform-quality) non-renewable resource. By assuming a single homogeneous stock, the possibility that substitute non-renewable resources exist is ruled out. The only substitution possibilities considered in Chapter 14 were between the non-renewable resource and other production inputs (labour and capital).

But in practice, non-renewable resources are heterogeneous. They comprise a set of resources varying in chemical and physical type (such as oil, gas, uranium, coal, and the various categories of each of these) and in terms of costs of extraction (as a result of differences in location, accessibility, quality and so on). This chapter investigates the efficient and optimal extraction of one component of this set of non-renewable resources where substitution possibilities exist. Substitution will take place if the price of the resource rises to such an extent that it makes alternatives economically more attractive. Consider, for example, the case of a country that has been exploiting its coal reserves, but in which coal extraction costs rise as lower-quality seams are mined. Meanwhile, gas costs fall as a result of the application of superior extraction and distribution technology. A point may be reached where electricity producers will substitute gas for coal in power generation. It is this kind of process that we wish to be able to model in this chapter.

Although the analysis that follows will employ a different (and in general, simpler) framework from that used in Chapter 14, one very important result carries over to the present case. The Hotelling rule is a necessary efficiency condition that must be satisfied by *any* optimal extraction programme. The chapter begins by laying out the conditions for the extraction path of a non-renewable resource stock to be socially optimal. It then considers how a resource is likely to be depleted in a market economy. As you would expect from the analysis in Chapters 4 and 11, the extraction path in competitive market economies will, under certain circumstances, be socially optimal. It is usually argued that one of these circumstances is that resource markets are competitive. We investigate this matter by comparing extraction paths under competitive and monopoly market structures against the benchmark of a 'first-best' social optimum.

The model used in most of this chapter is simple, and abstracts considerably from specific detail. The assumptions are gradually relaxed to deal with increasingly complex situations. To help understanding, it is convenient to begin with a model in which only two periods of time are considered. Even from such a simple starting point, very powerful results can be obtained, which can be generalised to analyses involving many periods. If you have a clear understanding of Hotelling's rule from Chapter 14, you might wish to skip the two-period model in the next section. Then, having analysed optimal depletion in a two-period model, a more general model is examined in which depletion takes place over T periods, where T may be a very large number.

There are two principal simplifications used in the chapter. First, we assume that utility comes directly from consuming the extracted resource. This is a considerably simpler, yet more specialised, case than that investigated in Chapter 14 where utility derived from consumption goods, obtained through a production function with a natural resource, physical capital (and, implicitly, labour) as inputs. Although doing this pushes the production function into the background, more attention is given to another kind of substitution possibility. As we remarked above, other non-renewable resources also exist. If one or more of these serve as substitutes for the resource being considered, this is likely to have important implications for economically efficient resource depletion paths.

Second, we do not take any account of adverse external effects arising from the extraction or consumption of the resource. The reader may find this rather surprising given that the production and consumption of non-renewable fossil-energy fuels are the primary cause of many of the world's most serious environmental problems. In particular, the combustion of these fuels accounts for 74% of carbon dioxide emissions in 2004 (IPCC, 2007a), at least 90% of sulphur dioxide in 2000 (Colls, 2002), and 67% of nitrogen oxide emissions in 1999 (IPCC, 2007b). In addition, fossil fuel use accounts for significant proportions of trace-metal emissions.

However, the relationship between non-renewable resource extraction over time and environmental degradation is so important that it warrants separate attention. This will be given in Chapter 16. Not surprisingly, we will show that the optimal extraction path will be different if adverse externalities are present causing environmental damage. The depletion model developed in this chapter will be used in Chapter 16 to derive some important results about efficient pollution targets and instruments.

Finally, a word about presentation. A lot of tedious – although not particularly difficult – mathematics is required to derive our results. The main text of this chapter lays emphasis on key results and the intuition which lies behind them; derivations, where they are lengthy, are placed in appendices. You may find it helpful to omit these on a first reading.

For much of the discussion in this chapter, it is assumed that there exists a known, finite stock of each kind of non-renewable resource. This assumption is not always appropriate. New discoveries are made, increasing the magnitude of known stocks, and technological change alters the proportion of mineral resources that are economically recoverable. Later sections indicate how the model may be extended to deal with some of these complications. Box 15.1 – which you should read now – considers several measures of resource stock, and throws some light on the issue of whether it can be reasonable to assume that there are fixed quantities of non-renewable resources.

Before reading the remainder of this chapter, the authors ask that you take great care when reading references to the price of the resource. There are several different meanings of that term. We have endeavoured to make clear which of these meanings is being used in each particular context. But you will find that the symbol P is not always being used in the same way in different places. We apologise for that inconsistency of notation.

Box 15.1 Are stocks of non-renewable resources fixed?

Non-renewable resources include a large variety of mineral deposits – in solid, liquid and gaseous forms – from which, often after some process of refining, metals, fossil fuels and other processed minerals are obtained. The crude forms of these resources are produced over very long periods of time by chemical, biological or physical processes. Their rate of formation is sufficiently slow in timescales relevant to humans that it is sensible to label such resources non-renewable. At any point in time, there exists some fixed, finite quantities of these resources in the earth's crust and environmental systems, albeit very large quantities in some cases.

So, in a physical sense, it is appropriate to describe non-renewable resources as existing in fixed quantities. However, that description may not be appropriate in an economic sense. To see why not, consider the information shown in Table 15.1. The final column – *Base resource* – indicates the mass of each resource that is thought to exist in the earth's crust. This is the measure closest to that we had in mind in the previous paragraph. However, most of this base resource consists of the mineral in very dispersed form, or at great depths below the surface. Base resource figures such as these are the broadest sense in which one might use the term 'resource stocks'. In each case, the measure is purely physical, having little or no relationship to economic measures of stocks. Notice that each of these quantities is extremely large relative to any other of the indicated stock measures.

The column labelled *Resource potential* is of more relevance to our discussions, comprising estimates of the upper limits on resource extraction possibilities given current and expected technologies. Whereas the base resource is a pure physical measure, the resource potential is a measure incorporating physical and technological information. But this illustrates the difficulty of classifying and measuring resources; as time passes, technology will almost certainly change, in ways that cannot be predicted today. As a result, estimates of the resource potential will change (usually rising) over time. To some writers, the possibility that resource constraints on economic activity will bite depends primarily on whether or not technological improvement in extracting usable materials from the huge stocks of base resources (thereby augmenting resource potential) will continue more-or-less indefinitely.

However, an economist is interested not in what is technically feasible but in what would become available under certain conditions. In other words, he or she is interested in resource supplies, or potential supplies. These will, of course, be shaped by physical and technological factors. But they will also depend upon resource market prices and the costs of extraction via their influence on exploration and research effort and on expected profitability. Data in the column labelled *World reserve base* consist of estimates of the upper bounds of resource stocks (including reserves that have not yet been discovered) that are economically recoverable under 'reasonable expectations' of future price, cost and technology possibilities. Those labelled *Reserves* consist of quantities that are economically recoverable under present configurations of costs and prices.

In economic modelling, the existence of fixed mineral resource stocks is often used as a simplifying assumption. But our observations suggest that we should be wary of this. In the longer term, economically relevant stocks are not fixed, and will vary with changing economic and technological circumstances.

Box 15.1 *continued*

Table 15.1 Production, consumption and reserves of some important resources: 2007, and selected figures for other years (figures in millions of metric tons)

Production		Reserves		World reserve base		Consumption	Resource potential	Base resource
		Quantity	Reserve life (yrs)	Reserve base	Base life (yrs)			Base resource (crustal mass)
Aluminium	22	23 000	100	28 000	>170	38	3 519 000	1 990 000 000 000
Iron ore	1900	150 000	79	340 000	179	1400	2 035 000	1 392 000 000 000
Potassium	na	8 400	Centuries	17 000	Centuries	26	na	408 000 000 000
Manganese	12.6	500	40	5 200	413	11.1	42 000	31 200 000 000
Phosphate	156	15 000	96	47 000	301	na	51 000	28 800 000 000
Fluorspar	5.7	230	40	470	82	na	20 000	10 800 000 000
Sulphur	66	na	na	5 000	66	57.5	600 000	9 600 000 000
Chromium	20	>475	>24	1 950	>98	na	3 260	2 600 000 000
Zinc	10.9	180	17	480	44	11.5	3 400	2 250 000 000
Nickel	1.66	70	42	150	90	1.47	2 590	2 130 000 000
Copper	15.60	490	31	940	60	18.50	2 120	1 510 000 000
Lead	3.77	79	21	170	45	6.3	550	290 000 000
Tin	0.333	5.6	17	11	33	0.367	68	40 000 000
Tungsten	0.0545	3	55	6.5	119	0.063	51	26 400 000
Mercury	0.0012	0.046	38	0.240	200	0.002	3.4	2 100 000
Silver	0.0208	0.27	13	0.57	27	0.028	2.8	1 800 000
Platinum group metals	0.00046	0.71	1543	0.80	1739	0.00042	1.2	1 100 000

Source: Data for 'Resource Potential' and 'Base Resource' are compiled from World Resources Institute publications, and refer to 1991 (but will remain largely unchanged today). Data on 'Production', 'Reserves' and 'World reserve base' are 2007 data, taken from US Geological Survey Minerals Information, online at http://minerals.usgs.gov/minerals/pubs/commodity/. Consumption data refer to 2007 unless stated otherwise, and are derived from various sources, including World Bureau of Metal Statistics and United Nations. Aluminium data refers to 1997, not including recycled production, online at http://www.un.org/esa/sustdev/publications/esa99dp5.pdf. Reserves refer to bauxite; reserve life and reserve base life are estimates by UNO. Consumption exceeds production because of recycling. Phosphate rock is the main source of potassium; figures in the last two columns refer to potassium rather than phosphate. Fluorspar is the main source of fluorine; figures in the last two columns refer to fluorine rather than fluorspar. Potassium data is from various sources, but not fully reliable.

15.1 A non-renewable resource two-period model

Consider a planning horizon that consists of two periods, period 0 and period 1. There is a fixed stock of known size of one type of a non-renewable resource. The initial stock of the resource (at the start of period 0) is denoted $\bar{S}$.[1] Let R_t be the quantity extracted in period t and assume that an inverse demand function exists for this resource at each time, given by

$$P_t = a - bR_t$$

where P_t is now referring to the gross (or market) price in period t, with a and b being positive constant numbers. So, the demand functions for the two periods will be:

$$P_0 = a - bR_0$$

$$P_1 = a - bR_1$$

These demands are illustrated in Figure 15.1.

A linear and negatively sloped demand function such as this one has the property that demand goes to zero at some price, in this case the price a. Hence, either this resource is non-essential or it possesses a substitute which at the price a becomes economically more attractive. The assumption of linearity of demand is arbitrary and so you should bear in mind that the particular results derived below are conditional upon the assumption that the demand curve is of this form.

The shaded area in Figure 15.1 (algebraically, the integral of P with respect to R over the interval $R = 0$ to $R = R_t$) shows the total benefit consumers obtain from consuming the quantity R_t in period t. From a social point of view, this area represents the gross social benefit, B, derived from the extraction and consumption of quantity R_t of the resource.[2] We can express this quantity as

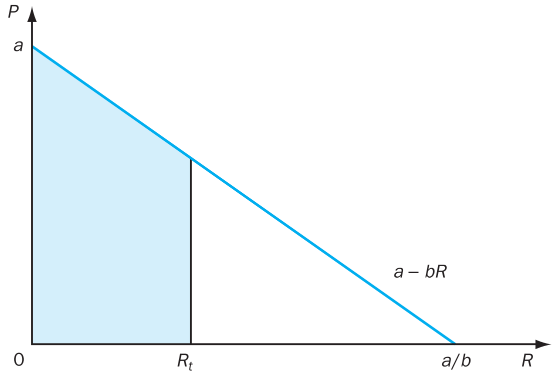


Figure 15.1 The non-renewable resource demand function for the two-period model

$$B(R_t) = \int_0^{R_t} (a - bR)\,\mathrm{d}R$$

$$= aR_t - \frac{b}{2}R_t^2$$

where the notation $B(R_t)$ is used to make explicit the fact that the gross benefit at time t (B_t) is dependent on the quantity of the resource extracted and consumed (R_t).

However, the gross benefit obtained by consumers is not identical to the net social benefit of the resource, as resource extraction involves costs. In this chapter, we assume that these costs are fully borne by the resource-extracting firms, and so private and social costs are identical.[3] This assumption will be dropped in the following chapter. Let us define c to be the constant marginal cost of extracting the resource ($c \geq 0$).[4] Then total extraction costs, C_t, for the extracted quantity R_t units will be

$$C_t = cR_t$$

[1] In the previous chapter, the initial quantity of the resource stock (at time period $t = 0$) was denoted by S_0. Here we are using $\bar{S}$ for the same quantity, and so there is some inconsistency of notation between Chapters 14 and 15.

[2] A demand curve is sometimes taken as providing information about the marginal willingness to pay (or marginal benefit) for successive units of the good in question. The area under a demand curve up to some given quantity is, then, the sum of a set of marginal benefits, and is equal to the total benefit derived from consuming that quantity.

[3] We also assume that benefits represented in the resource demand function are the only benefits to society, so there are no beneficial externalities.

[4] Constancy of marginal costs of extraction is a very strong assumption. In the previous chapter, we investigated a more general case in which marginal extraction costs are not necessarily constant. We do not consider this any further here. Later in this chapter, however, we do analyse the consequences for extraction of a once-and-for-all rise in extraction costs.

The total net social benefit from extracting the quantity R_t is

$$\text{NSB}_t = B_t - C_t$$

where NSB denotes the total net social benefit and B is the gross social benefit of resource extraction and use.[5] Hence

$$\text{NSB}(R_t) = \int_0^{R_t} (a - bR)\,\text{d}R - cR_t = aR_t - \frac{b}{2}R_t^2 - cR_t \tag{15.1}$$

15.1.1 A socially optimal extraction policy

Our objective in this subsection is to identify a socially optimal extraction programme. This will serve as a benchmark in terms of which any particular extraction programme can be assessed. In order to find the socially optimal extraction programme, two things are required. The first is a social welfare function that embodies society's objectives; the second is a statement of the technical possibilities and constraints available at any point in time. Let us deal first with the social welfare function, relating this as far as possible to our discussion of social welfare functions in Chapters 3 and 4.

As in Chapter 3, the social welfare function that we shall use is discounted utilitarian in form. So the general two-period social welfare function

$$W = W(U_0, U_1)$$

takes the particular form

$$W = U_0 + \frac{U_1}{1 + \rho}$$

where ρ is the social utility discount rate, reflecting society's time preference. Now regard the utility in each period as being equal to the net social benefit in each period.[6] Given this, the social welfare function may be written as

$$W = \text{NSB}_0 + \frac{\text{NSB}}{1 + \rho}$$

Only one relevant technical constraint exists in this case: there is a fixed initial stock of the non-renewable resource, $\bar{S}$. We assume that society wishes to have none of this resource stock left at the end of the second period. Then the quantities extracted in the two periods, R_0 and R_1, must satisfy the constraint:[7]

$$R_0 + R_1 = \bar{S}$$

The optimisation problem can now be stated. Resource extraction levels R_0 and R_1 should be chosen to maximise social welfare, W, subject to the constraint that total extraction of the resources over the two periods equals $\bar{S}$. Mathematically, this can be written as

$$\max_{R_0,\, R_1} W = \text{NSB}_0 + \frac{\text{NSB}}{1 + \rho}$$

subject to

$$R_0 + R_1 = \bar{S}$$

There are several ways of obtaining solutions to constrained optimisation problems of this form. We use the Lagrange multiplier method, a technique that was explained in Appendix 3.1. The first step is to form the Lagrangian function, L:

$$\begin{aligned} L = W - \lambda(\bar{S} - R_0 - R_1) &= (\text{NSB}_0) + \left(\frac{\text{NSB}_1}{1 + \rho}\right) \\ - \lambda(\bar{S} - R_0 - R_1) &= \left(aR_0 - \frac{b}{2}R_0^2 - cR_0\right) \\ &+ \left(\frac{aR_1 - \frac{b}{2}R_1^2 - cR_1}{1 + \rho}\right) - \lambda(\bar{S} - R_0 - R_1) \end{aligned} \tag{15.2}$$

in which λ is a 'Lagrange multiplier'. Remembering that R_0 and R_1 are choice variables – variables whose

[5] Strictly speaking, social benefits derive from consumption (use) of the resource, not extraction *per se*. However, we assume throughout this chapter that all resource stocks extracted in a period are consumed in that period, and so this distinction becomes irrelevant.

[6] In order to make such an interpretation valid, we shall assume that the demand function is 'quasi-linear' (see Varian, 1987). Suppose there are two goods, X, the good whose demand we are interested in, and Y, money to be spent on all other goods. Quasi-linearity requires that the utility function for good X be of the form $U = V(X) + Y$. This implies that income effects are absent in the sense that changes in income do not affect the demand for good X. In this case, we can legitimately interpret the area under the demand curve for good X as a measure of utility.

[7] The problem could easily be changed so that a predetermined quantity S^* ($S^* \geq 0$) must be left at the end of period 1 by rewriting the constraint as $R_0 + R_1 + S^* = \bar{S}$. This would not alter the essence of the conclusion we shall reach.

value must be selected to maximise welfare – the necessary conditions include:

$$\frac{\partial L}{\partial R_0} = a - bR_0 - c + \lambda = 0 \tag{15.3}$$

$$\frac{\partial L}{\partial R_1} = \frac{a - bR_1 - c}{1 + \rho} + \lambda = 0 \tag{15.4}$$

Since the right-hand side terms of equations 15.3 and 15.4 are both equal to zero, this implies that

$$a - bR_0 - c = \frac{a - bR_1 - c}{1 + \rho}$$

Using the demand function $P_t = a - bR_t$, the last equation can be written as

$$P_0 - c = \frac{P_1 - c}{1 + \rho}$$

where P_0 and P_1 are gross prices and $P_0 - c$ and $P_1 - c$ are net prices. A resource's net price is also known as the resource rent or resource royalty. Rearranging this expression, we obtain

$$\rho = \frac{(P_1 - c) - (P_0 - c)}{(P_0 - c)}$$

If we change the notation used for time periods so that $P_0 = P_{t-1}$, $P_1 = P_t$ and $c = c_t = c_{t-1}$, we then obtain

$$\rho = \frac{(P_t - c_t) - (P_{t-1} - c_{t-1})}{(P_{t-1} - c_{t-1})} \tag{15.5}$$

which is equivalent to a result we obtained previously in Chapter 14, equation 14.15, commonly known as Hotelling's rule. Note that in equation 15.5, P is a gross price whereas in equation 14.15, P refers to a net price, resource rent or royalty. However, since $P - c$ in equation 15.5 is the resource net price or royalty, these two equations are identical (except for the fact that one is in discrete-time notation and the other in continuous-time notation).

What does this result tell us? The left-hand side of equation 15.5, ρ, is the social utility discount rate, which embodies some view about how future utility should be valued in terms of present utility. The right-hand side is the proportionate rate of growth of the resource's net price. So if, for example, society chooses a discount rate of 0.1 (or 10%), Hotelling's rule states that an efficient extraction programme requires the net price of the resource to grow at a proportionate rate of 0.1 (or 10%) over time.

Now we know how much higher the net price should be in period 1 compared with period 0, if welfare is to be maximised; but what should be the level of the net price in period 0? This is easily answered. Recall that the economy has some fixed stock of the resource that is to be entirely extracted and consumed in the two periods. Also, we have assumed that the demand function for the resource is known. An optimal extraction programme requires two gross prices, P_0 and P_1, such that the following conditions are satisfied:

$$P_0 = a - bR_0$$

$$P_1 = a - bR_1$$

$$R_0 + R_1 = \bar{S}$$

$$P_1 - c = (1 + \rho)(P_0 - c)$$

This will uniquely define the two prices (and so the two quantities of resources to be extracted) that are required for welfare maximisation. Problem 1, at the end of this chapter, provides a numerical example to illustrate this kind of two-period optimal depletion problem. You are recommended to work through this problem before moving on to the next section.

15.2 A non-renewable resource multi-period model

Having investigated resource depletion in the simple two-period model, the analysis is now generalised to many periods. It will be convenient to change from a discrete-time framework (in which there is a number of successive intervals of time, denoted period 0, period 1, etc.) to a continuous-time framework which deals with rates of extraction and use at particular points in time over some continuous-time horizon.[8]

To keep the maths as simple as possible, we will push extraction costs somewhat into the background. To do this, P is now defined to be the net price of the non-renewable resource, that is, the price after deduction of the cost of extraction. Let $P(R)$ denote

[8] The material in this section, in particular the worked example investigated later, owes much to Heijman (1990).

the inverse demand function for the resource, indicating that the resource **net** price is a function of the quantity extracted, R. The social utility from consuming a quantity R of the resource may be defined as

$$U(R) = \int_0^R P(R)\,\mathrm{d}R \tag{15.6a}$$

which is illustrated by the shaded area in Figure 15.2. You will notice that the demand curve used in Figure 15.2 is non-linear. We shall have more to say about this particular form of the demand function shortly.

By differentiating total utility with respect to R, the rate of resource extraction and use, we obtain

$$\frac{\partial U}{\partial R} = P(R) \tag{15.6b}$$

which states that the marginal social utility of resource use equals the **net** price of the resource.

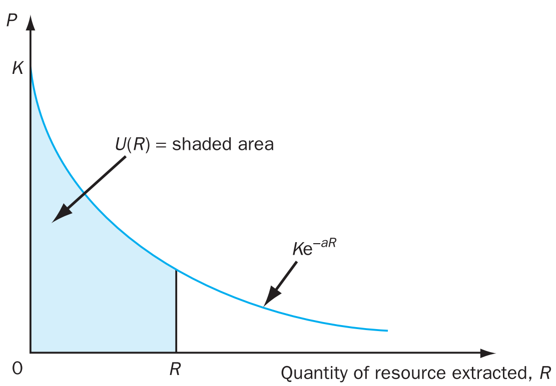


Figure 15.2 A resource demand curve, and the total utility from consuming a particular quantity of the resource

Assume, as for the two-period model, that the intertemporal social welfare function is utilitarian. Future utility is discounted at the instantaneous social utility discount rate ρ. Then the value of social welfare over an interval of time from period 0 to period T can be expressed as[9]

$$W = \int_0^T U(R_t)\mathrm{e}^{-\rho t}\mathrm{d}t$$

Our problem is to make social-welfare-maximising choices of

(a) R_t, for $t = 0$ to $t = T$ (that is, we wish to choose a quantity of resource to be extracted in each period), and
(b) the optimal value for T (the point in time at which depletion of the resource stock ceases), subject to the constraint that

$$\int_0^T R_t\mathrm{d}t = \bar{S}$$

where $\bar{S}$ is the total initial stock of the non-renewable resource. That is, the total extraction of the resource is equal to the size of the initial resource stock. Note that in this problem, the time horizon to exhaustion is being treated as an endogenous variable to be chosen by the decision maker.

We define the remaining stock of the natural resource at time t, S_t, as

$$S_t = \bar{S} - \int_{\tau=0}^{\tau=t} R_\tau\mathrm{d}\tau$$

then by differentiation with respect to time, we obtain

[9] It may be helpful to relate this form of social welfare function to the discrete-time versions we have been using previously. We have stated that a T-period discrete-time discounted welfare function can be written as

$$W = U_0 + \frac{U_1}{1+\rho} + \frac{U_2}{(1+\rho)^2} + \ldots + \frac{U_r}{(1+\rho)^r}$$

We could write this equivalently as

$$W = \sum_{t=0}^{t=T} \frac{U_t}{(1+\rho)^T}$$

A continuous-time analogue of this welfare function is then

$$W = \int_{t=0}^{t=T} U_t\mathrm{e}^{-\rho t}\mathrm{d}t$$

$$\dot{S}_t = -R_t$$

where $\dot{S} = \mathrm{d}S/\mathrm{d}t$, the rate of change of the remaining resource stock with respect to time.

So the dynamic optimisation problem involves the choice of a path of resource extraction R_t over the interval $t = 0$ to $t = T$ that satisfies the resource stock constraint and which maximises social welfare, W. Mathematically, we have:

$$\text{Max } W = \int_0^T U(R_t)\mathrm{e}^{-\rho t}\mathrm{d}t$$

subject to $\dot{S}_t = -R_t$

It would be a useful exercise at this point for you to use the optimisation technique explained in Appendix 14.1 to derive the solution to this problem. Your derivation can be checked against the answer given in Appendix 15.1.

Thinking point

Before moving on to interpret the main components of this solution, it will be useful to pause for a moment to reflect on the nature of this model. It is similar in general form to the model we investigated in Chapter 14, and laid out in full in Appendix 14.2. However, the model is simpler in one important way from that of the previous chapter as utility is derived directly from the consumption of the natural resource, rather than indirectly from the consumption goods generated through a production function. There is a fixed, total stock available of the natural resource, and this model is sometimes called the 'cake-eating' model of resource depletion.

It would also be reasonable to interpret this model as one in which a production function exists implicitly. However, this production function has just one argument – the non-renewable natural resource input – as compared with the two arguments – the natural resource and human-made capital – in the model of Chapter 14.

It is clear that this model can at most be regarded as a partial account of economic activity. One possible interpretation of this partial status is that the economy also produces, or could produce, goods and services through other production functions, using capital, labour and perhaps renewable resource inputs. In this interpretation the non-renewable resource is like a once-and-for-all gift of nature. Using this non-renewable resource provides something over and above the welfare possible from production in its absence. It is this *additional welfare* that is being measured by our term W.

An alternative interpretation is more commonly found in the literature. Here, non-renewable resources consist of a diverse set of different resources. Each element of this set is a particular resource that is fixed and homogeneous. Substitution possibilities exist between at least some elements of this set of resources. For historical, technical or economic reasons, production might currently rely particularly heavily on one kind of resource. Changing technological or economic conditions might lead to this stock being replaced by another. With the passage of time, a sequence of resource stocks are brought into play, with each one eventually being replaced by another. In this story, what our resource depletion model investigates is one stage in this sequence of depletion processes. This interpretation will be used later in the chapter when the concepts of a backstop technology and a choke price are introduced.

These comments raise a general issue about choices that need to be made in doing resource modelling. It is often too difficult to explain everything of interest in one framework. Sometimes, one needs to pick 'horses for courses'. In the previous chapter, we were concerned with substitution between natural resources and physical capital; that required that we explicitly specify a conventional type of production function. In this chapter, that is not of central concern, and so the production function can be allowed to slip somewhat into the background. However, we do wish here to place emphasis on substitution processes between natural resources. That can be done in a simple way, by paying greater attention to the nature of resource demand functions, and to the idea of a choke price for a resource.

Whether or not you have succeeded in obtaining a formal solution to this optimisation problem, intuition should suggest one condition that must be satisfied if W is to be maximised. R_t must be chosen so that the *discounted* marginal utility is equal at each point in time, that is,

$$\frac{\partial U}{\partial R}e^{-\rho t} = \text{constant}$$

To understand this, let us use the method of contradiction. If the discounted marginal utilities from resource extraction were not equal in every period, then total welfare W could be increased by shifting some extraction from a period with a relatively low discounted marginal utility to a period with a relatively high discounted marginal utility. Rearranging the path of extraction in this way would raise welfare. It must, therefore, be the case that welfare can only be maximised when discounted marginal utilities are equal. What follows from this result? First note equation 15.6b again:

$$\frac{\partial U_t}{\partial R_t} = P_t$$

So, the requirement that the discounted marginal utility be constant is equivalent to the requirement that the discounted net price is constant as well – a result noted previously in Chapter 14. That is,

$$\frac{\partial U_t}{\partial R_t}e^{-\rho t} = P_t e^{-\rho t} = \text{constant} = P_0$$

Rearranging this condition, we obtain

$$P_t = P_0 e^{\rho t} \tag{15.7a}$$

By differentiation[10] this can be rewritten as

$$\frac{\dot{P}_t}{P_t} = \rho \tag{15.7b}$$

This is, once again, the Hotelling efficiency rule. It now appears in a different guise, because of our switch to a continuous-time framework. The rule states that the net price or royalty P_t of a non-renewable resource should rise at a rate equal to the social utility discount rate, ρ, if the social value of the resource is to be maximised.

We now know the rate at which the resource net price or royalty must rise. However, this does not fully characterise the solution to our optimising problem. There are several other things we need to know too. First, we need to know the optimal initial value of the resource net price. Secondly, we need to know over how long a period of time the resource should be extracted – in other words, what is the optimal value of T? Thirdly, what is the optimal rate of resource extraction at each point in time? Finally, what should be the values of P and R at the end of the extraction horizon?

It is not possible to obtain answers to these questions without one additional piece of information: the particular form of the resource demand function. So let us suppose that the resource demand function is

$$P(R) = Ke^{-aR} \tag{15.8}$$

which is illustrated in Figure 15.2.[11] Unlike the demand function used in the two-period analysis, this function exhibits a non-linear relationship between P and R, and is probably more representative of the form that resource demands are likely to take than the linear function used in the section on the two-period model. However, it is similar to the

[10] Differentiation of equation 15.7a with respect to time gives

$$dP_t/dt \equiv \dot{P}_t = P_0\rho e^{\rho t}$$

By substitution of equation 15.7a into this expression, we obtain

$$\dot{P}_t = \rho P_t$$

and dividing through by P_t we obtain

$$\dot{P}_t/P_t = \rho$$

as required.

[11] For the demand function given in equation 15.8, we can obtain the particular form of the social welfare function as follows. The social utility function corresponding to equation 15.6a will be:

$$U(R) = \int_0^R P(R)dR = \int_0^R Ke^{-aR}dR = \frac{k}{a}(1 - e^{-aR})$$

The social welfare function, therefore, is

$$W = \int_0^T U(R_t)e^{-\rho t}dt = \int_0^T \frac{K}{a}(1 - e^{-aR})e^{-\rho t}dt$$

previous demand function in so far as it exhibits zero demand at some finite price level. To see this, just note that $P(R = 0) = K$. K is the so-called *choke price* for this resource, meaning that the demand for the resource is driven to zero or is 'choked off' at this price. At the choke price people using the services of this resource would switch demand to some alternative, substitute, non-renewable resource, or to an alternative final product not using that resource as an input.

As we shall demonstrate shortly, given knowledge of

- a particular resource demand function,
- Hotelling's efficiency condition,
- an initial value for the resource stock, and
- a final value for the resource stock,

it is possible to obtain expressions for the optimal initial, interim and final resource net price (royalty) and resource extraction rates. What about the final stock level? This is straightforward. An optimal solution must have the property that the stock goes to zero at exactly the same point in time that demand and extraction go to zero.[12] If that were not the case, some resource will have been needlessly wasted. So we know that the solution must include $S_T = 0$ and $R_T = 0$, with resource stocks being positive, and positive extraction taking place over all time up to T. As you will see below, that will give us sufficient information to fully tie down the solution.

Before we proceed to obtain all the details of the solution, one important matter must be reiterated. The solution to a problem of this type will depend upon the demand function chosen. Hence the particular solutions derived below are conditional upon the demand function chosen, and will not be valid in all circumstances. Our model in this chapter assumes that the resource has a choke price, implying that a substitute for the resource becomes economically more attractive at that price. If you wish to examine the case in which there is no choke price – indeed, where there is no finite upper limit on the resource price – you may find it useful to work through some of the exercises provided in the *Additional Materials* for this chapter, which deal with this case among others.

Table 15.2 Optimality conditions for the multi-period model

	Initial ($t = 0$)	Interim ($t = t$)	Final ($t = T$)
Royalty, P	$P_0 = Ke^{-\sqrt{2\rho\bar{S}a}}$	$P_t = Ke^{\rho\{t-T\}}$	$P_T = K$
Extraction, R	$R_0 = \sqrt{\dfrac{2\rho\bar{S}}{a}}$	$R_t = \dfrac{\rho}{a}(T-t)$	$R_T = 0$
Depletion time			$T = \sqrt{\dfrac{2\bar{S}a}{\rho}}$

As the mathematics required to obtain the full solution are rather tedious (but not particularly difficult), the derivations are presented in Appendix 15.1. You are strongly recommended to read this now, but if you prefer to omit these derivations, the results are presented in Table 15.2. There it can be seen that all the expressions for the initial, interim and final resource royalty (or net prices) and rate of resource extraction are functions of the parameters of the model (K, ρ and a) and T, the optimal depletion time. As the final expression indicates, T is itself a function of those parameters. Given the functional forms we have been using in this section, if the values of the parameters K, ρ and a were known, it would be possible to solve the model to obtain numerical values for all the variables of interest over the whole period for which the resource will be extracted.

Figure 15.3 portrays the solution to our optimal depletion model. The diagram shows the optimal resource extraction and net price paths over time corresponding to social welfare maximisation. As we show subsequently, it also represents the profit-maximising extraction and price paths in perfectly competitive markets. In the upper right quadrant, the net price is shown rising exponentially at the social utility discount rate, ρ, thereby satisfying the Hotelling rule. The upper left quadrant shows the resource demand curve with a choke price K. The lower left quadrant gives the optimal extraction path of the non-renewable resource, which is, in this case, a linear declining function of time.

[12] In terms of optimisation theory, this constitutes a so-called terminal condition for the problem.

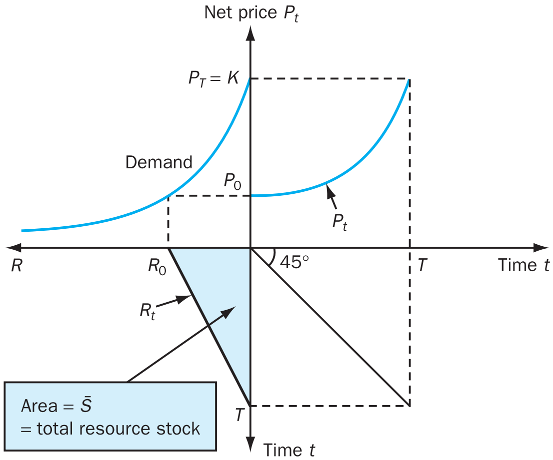


Figure 15.3 Graphical representation of solutions to the optimal resource depletion model

The net price is initially at P_0, and then grows until it reaches the choke price K at time T. At this point, demand for the resource goes to zero, and the accumulated extraction of the resource (the shaded area beneath the extraction path) is exactly equal to the total initial resource stock, $\bar{S}$. The lower right quadrant maps the time axes by a 45° line. A worked numerical example illustrating optimal extraction is presented in Appendix 15.3.

15.3 Non-renewable resource extraction in perfectly competitive markets

Until this point, we have said nothing about the kind of market structure in which decisions are made. It is as if we have been imagining that a rational social planner were asked to make decisions that maximise social welfare, given the constraints facing the economy. The optimality conditions listed in Table 15.2, plus the Hotelling efficiency condition, are the outcome of the social planner's calculations. How will matters turn out if decisions are instead the outcome of profit-maximising decisions in a perfectly competitive market economy? This section demonstrates that, *ceteris paribus*, the outcomes will be identical. Hotelling's rule and the optimality conditions of Table 15.2 are also obtained under a perfect competition assumption.

Suppose there are m competitive firms in the market. Use the subscript j to denote any one of these m firms. Assume, for simplicity, that all firms have equal and constant marginal costs of extracting the resource. Now as all firms in a competitive market face the same fixed selling price at any point in time, the market royalty will be identical over firms. Given the market royalty P_t, each firm chooses an amount to extract and sell, $R_{j,t}$, to maximise its profits. Mathematically, the jth firm's objective is to maximise

$$\int_0^T \Pi_{j,t} e^{-it} dt$$

subject to

$$\int_0^T \left(\sum_{j=1}^m R_{j,t}\right) dt = \bar{S}$$

where $\Pi_j = P \cdot R_j$ is firm j's profit and i is the market interest rate. Note that the same stock constraint operates on all firms collectively; the industry as a whole cannot extract more than the fixed initial stock over the whole time horizon. The profit-maximising extraction path is obtained when each firm selects an extraction $R_{j,t}$ at each time, $t = 0$ to $t = T$, so that its discounted marginal profit will be the same at any point in time t, that is,

$$M\Pi_{j,t} e^{-it} = \frac{\partial \Pi_{j,t}}{\partial R_{j,t}} e^{-it} = \frac{\partial P R_{j,t}}{\partial R_{j,t}} e^{-it} = P_t e^{-it}$$
$$= \text{constant, for } t = 0 \text{ to } t = T$$

where $M\Pi_j$ is firm j's marginal profit function. If discounted marginal profits were *not* the same over time, total profits could be increased by switching extraction between time periods so that more was extracted when discounted profits were high and less when they were low. The result that the discounted marginal profit is the same at any point in time implies that

$$P_t e^{-it} = P_0 \text{ or } P_t = P_0 e^{it}$$

Not surprisingly, Hotelling's efficiency rule continues to be a required condition for profit maximisation, so that the market net price of the resource must grow over time at the rate i. The interest rate in this profit maximisation condition is the market rate of interest. Our analysis in Chapter 11 showed that, in perfectly competitive capital markets and in the absence of transactions costs, the market interest rate will be equal to r, the consumption rate of interest, and also to δ, the rate of return on capital.

We appear now to have two different efficiency conditions,

$$\frac{\dot{P}}{P} = \rho \text{ and } \frac{\dot{P}}{P} = i$$

the former emerging from maximising social welfare, the latter from private profit maximisation. But these are in fact identical conditions under the assumptions we have made in this chapter; by assuming that we can interpret areas under demand curves (that is, gross benefits) as quantities of utility, we in effect impose the condition that $\rho = r$. Given this result, it is not difficult to show, by cranking through the appropriate maths in a similar manner to that done in Appendix 15.1, that all the results of Table 15.2 would once again be produced under perfect competition, **provided** the private market interest rate equals the social consumption discount rate. We leave this as an exercise for the reader.

Finally, note that the appearance of a positive net price or royalty, $P_t > 0$, for non-renewable resources reflects the fixed stock assumption. If the resource existed in unlimited quantities (that is, the resource were not scarce) net prices would be zero in perfect competition, as the price of the product will equal the marginal cost (c), a result which you may recall from standard theory of long-run equilibrium in competitive markets. In other words, scarcity rent would be zero as there would be no scarcity.

15.4 Resource extraction in a monopolistic market

It is usual to assume that the objective of a monopoly is to maximise its discounted profit over time. Thus, it selects the net price P_t (or royalty) and chooses the output R_t so as to maximise

$$\int_0^T \Pi_t e^{-it} dt$$

subject to

$$\int_0^T R_t dt = \bar{S}$$

where $\Pi_t = P(R_t)R_t$.

For the same reason as in the case of perfect competition, the profit-maximising solution is obtained by choosing a path for R so that the discounted marginal profit will be the same at any time. So we have

$$M\Pi_t e^{-it} = \frac{\partial \Pi_t}{\partial R_t} e^{-it} = \text{constant} = M\Pi_0$$

that is,

$$M\Pi_t = M\Pi_0 e^{it} \tag{15.9}$$

Looking carefully at equation 15.9, and comparing this with the equation for marginal profits in the previous section, it is clear why the profit-maximising solutions in monopolistic and competitive markets will differ. Under perfect competition, the market price is exogenous to (fixed for) each firm. Thus we are able to obtain the result that in competitive markets, marginal revenue equals price. However, in a monopolistic market, price is not fixed, but will depend upon the firm's output choice. Marginal revenue will be less than price in this case.

The necessary condition for profit maximisation in a monopolistic market states that the marginal profit (and not the net price or royalty) should increase at the rate of interest i in order to maximise the discounted profits over time. The solution to the monopolist's optimising problem is derived in Appendix 15.2. If you wish to omit this, you will find the results in Table 15.3.

15.5 A comparison of competitive and monopolistic extraction programmes

Table 15.3 summarises the results concerning optimal resource extraction in perfectly competitive and monopolistic markets. The analytical results presented are derived in Appendices 15.1 and 15.2.

Table 15.3 The comparison table: perfect competition v. monopoly

	Perfect competition	Monopoly
Objective	$\max \int_0^T P_t R_t^j e^{-it} dt$	$\max \int_0^T P(R_t) R_t e^{-it} dt$
Constraint	$\int_0^T \left(\sum_j R_t^j \right) dt = \bar{S}$	$\int_0^T R_t dt = \bar{S}$
Demand curve	$P_t = K e^{-aR_t}$	$P_t = K e^{-aR_t}$
		Optimal solution
Exhaustion time	$T = \sqrt{\frac{2\bar{S}a}{i}}$	$T = \sqrt{\frac{2\bar{S}ah}{i}}$
Initial royalty	$P_0 = K e^{-\sqrt{2i\bar{S}a}}$	$P_0 = K e^{-\sqrt{\frac{2i\bar{S}a}{i}}}$
Royalty path	$P_t = P_0 e^{it}$	$P_t = P_0 e^{(it/h)}$
Extraction path	$R_t = \frac{i}{a}(T - t)$	$R_t = \frac{i}{ha}(T - t)$
	where $R_i = \sum_j R_i^j$	$R_i = \sum_j R_i^j$
	$R_0 = \sqrt{\frac{2i\bar{S}}{a}}$	$R_0 = \sqrt{\frac{2i\bar{S}}{ha}}$

P_t is the net price (royalty) of non-renewable resource with fixed stock $\bar{S}$
R_t is the total extraction of the resource at time t
R_t^j is the extraction of individual firm j at time t
i is the interest rate
T is the exhaustion time of the natural resource
K and a are fixed parameters
$h = (1.6)^2$

For convenience, the notation used in Table 15.3 is listed above.

Two key results emerge from Tables 15.2 and 15.3. First, under certain conditions, there is equivalence between the perfect competition market outcome and the social welfare optimum. If all markets are perfectly competitive, and the market interest rate is equal to the social consumption discount rate, the profit-maximising resource depletion programme will be identical to the one that is socially optimal.

Secondly, there is non-equivalence of perfect competition and monopoly markets: profit-maximising extraction programmes will be different in perfectly competitive and monopolistic resource markets.

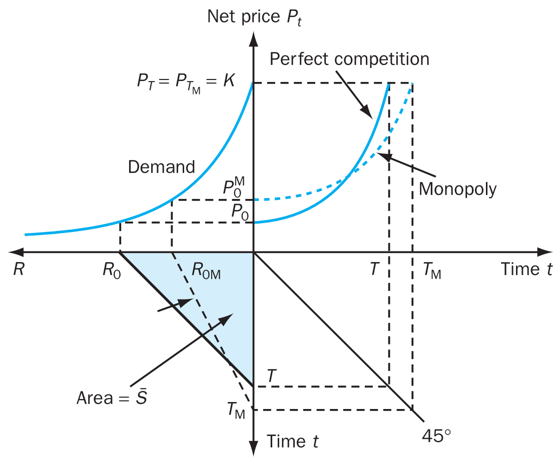


Figure 15.4 A comparison of resource depletion in competitive and monopolistic markets

Given the result stated in the previous paragraph, this implies that monopoly must be sub-optimal in a social-welfare-maximising sense.

For the functional forms we have used in this section, a monopolistic firm will take $\sqrt{h} = 1.6$ times longer to fully deplete the non-renewable resource than a perfectly competitive market in our model. As Figure 15.4 demonstrates, the initial net price will be higher in monopolistic markets, and the rate of price increase will be slower. Extraction of the resource will be slower at first in monopolistic markets, but faster towards the end of the depletion horizon. Monopoly, in this case at least, turns out to be an ally of the conservationist, in so far as the time until complete exhaustion is deferred further into the future.[13] As the comparison in Figure 15.4 illustrates, a monopolist will restrict output and raise prices initially, relative to the case of perfect competition. The rate of price increase, however, will be slower than under perfect competition. Eventually, an effect of monopolistic markets is to increase the time horizon over which the resource is extracted. We illustrate these results numerically in the Excel file *polcos.xls*, the contents of which are explained in the Word file *polcos.doc*. These can both be found in the *Additional Materials* for Chapter 15.

[13] Note that this conclusion is not necessarily the case. The longer depletion period we have found is a consequence of the particular assumptions made here. Although in most cases one would expect this to be true, it is possible to make a set of assumptions such that a monopolist would extract the stock in a shorter period of time.

15.6 Extensions of the multi-period model of non-renewable resource depletion

To this point, a number of simplifying assumptions in developing and analysing our model of resource depletion have been made. In particular, it has been assumed that

- the utility discount rate and the market interest rate are constant over time;
- there is a fixed stock, of known size, of the non-renewable natural resource;
- the demand curve is identical at each point in time;
- no taxation or subsidy is applied to the extraction or use of the resource;
- marginal extraction costs are constant;
- there is a fixed 'choke price' (hence implying the existence of a backstop technology);
- no technological change occurs;
- no externalities are generated in the extraction or use of the resource.

We shall now undertake some comparative dynamic analysis. This consists of finding how the optimal paths of the variables of interest change over time in response to changes in the levels of one or more of the parameters in the model, or of finding how the optimal paths alter as our assumptions are changed. We adopt the device of investigating changes to one parameter, holding all others unchanged, comparing the new optimal paths with those derived above for our simple multi-period model. (We shall only discuss these generalisations for the case of perfect competition; analysis of the monopoly case is left to the reader as an exercise.)

The reader interested in doing comparative dynamics analysis by Excel simulation may wish to explore the file *hmodel.xls* (together with its explanatory document, *hmodel.doc*) in the *Additional Materials* to Chapter 15. The consequences of each of the changes described in the following subsections can be verified using that Excel workbook.

15.6.1 An increase in the interest rate

Let us make clear the problem we wish to answer here. Suppose that the interest rate we had assumed in drawing Figure 15.3 was 6% per year. Now suppose that the interest rate was not 6% but rather 10%; how would Figure 15.3 have been different if the interest rate had been higher in this way? This is the kind of question we are trying to answer in doing comparative dynamics.

The answer is shown in Figure 15.5. The thick, heavily drawn line represents the original optimal price path, with the price rising from an initial level of P_0 to its choke price, K, at time T. Now suppose that the interest rate rises. Since the resource's net price must grow at the market interest rate, an increase in i will raise the growth rate of the resource royalty, P_t; hence the new price path must have a steeper slope than the original one. The new price path will be the one labelled C in Figure 15.5. It will have an initial price lower than the one on the original price path, will grow more quickly, and will reach its final (choke) price earlier in time (before $t = T$). This result can be explained by the following observations. First, the choke price itself, K, is not altered by the interest rate change. Second, as we have already observed, the new price path must rise more steeply with a higher interest rate. Third, we can deduce that it must begin from a lower initial price level from using the resource exhaustion constraint. The change in interest rate does not alter the quantity that is to be extracted; the same total stock is extracted whatever the interest rate might be. If the price path began from the same initial value (P_0)

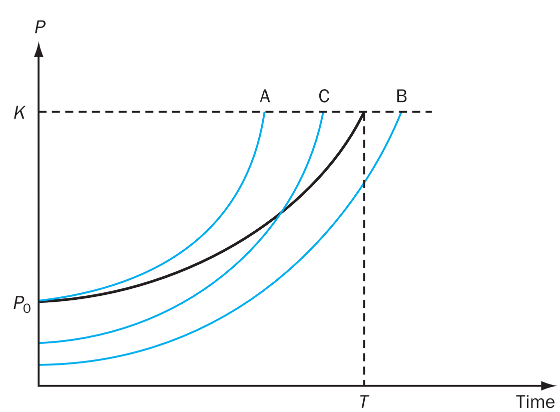


Figure 15.5 The effect of an increase in the interest rate on the optimal price of the non-renewable resource

then it would follow a path such as that shown by the curve labelled A and would reach its choke price before $t = T$. But then the price would always be higher than along the original price path, but for a shorter period of time. Hence the resource stock will not be fully extracted along path A and that path could not be optimal.

A path such as B is not feasible. Here the price is always lower (and so the quantity extracted is higher) than on the original optimal path, and for a longer time. But that would imply that more resources are extracted over the life of the resource than were initially available. This is not feasible. The only feasible and optimal path is one such as C. Here the price is lower than on the original optimal path for some time (and so the quantity extracted is greater); then the new price path crosses over the original one and the price is higher thereafter (and so the quantity extracted is lower).

Note that because the new path must intersect the original path from below, the optimal depletion time will be shorter for a higher interest rate. This is intuitively reasonable. Higher interest rate means greater impatience. More is extracted early on, less later, and total time to full exhaustion is quicker. The implications for all the variables of interest, P_t, T and R, are summarised in Figure 15.6.

15.6.2 An increase in the size of the known resource stock

In practice, estimates of the size of reserves of non-renewable resources such as coal and oil are under constant revision. *Proven reserves* are those unextracted stocks known to exist and can be recovered at current prices and costs. *Probable reserves* are stocks that are known, with near certainty, to exist but which have not yet been fully explored or researched. They represent the best guess of additional amounts that could be recovered at current price and cost levels. *Possible reserves* are stocks in geological structures near to proven fields. As prices rise, what were previously uneconomic stocks become economically recoverable.

Consider the case of a single new discovery of a fossil fuel stock. Other things being unchanged, if the royalty path were such that its initial level remained unchanged at P_0, then given the fact that the rate of royalty increase is unchanged, some proportion of the reserve would remain unutilised by the time the choke price, K, is reached. This is clearly neither efficient nor optimal. It follows that the initial royalty must be lower and the time to exhaustion is extended. At the time the choke price is reached, T', the new enlarged resource stock will have just reached complete exhaustion, as shown in Figure 15.7.

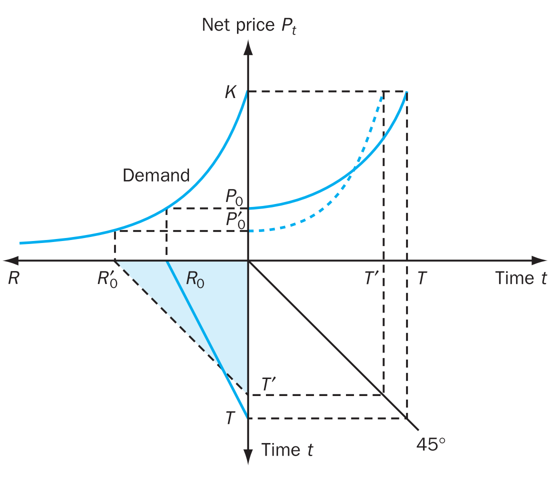


Figure 15.6 An increase in interest rates in a perfectly competitive market

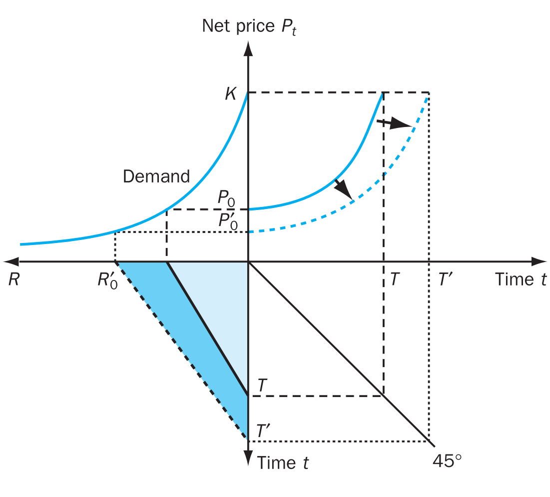


Figure 15.7 An increase in the resource stock

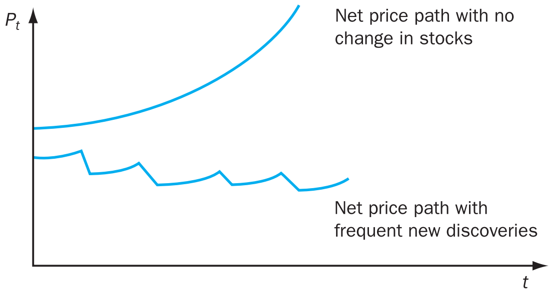


Figure 15.8 The effect of frequent new discoveries on the resource net price or royalty

Now suppose that there is a sequence of new discoveries taking place over time, so that the size of known reserves increases in a series of discrete steps. Generalising the previous argument, we would expect the behaviour of the net price or royalty over time to follow a path similar to that illustrated in Figure 15.8. This hypothetical price path is one that is consistent with the actual behaviour of oil prices.

15.6.3 Changing demand

Suppose that there is an increase in demand for the resource, possibly as a result of population growth or rising real incomes. The demand curve thus shifts outwards. Given this change, the old royalty or net price path would result in higher extraction levels, which will exhaust the resource before the net price has reached *K*, the choke price. Hence the net price must increase to dampen down quantities demanded; as Figure 15.9 shows, the time until the resource stock is fully exhausted will also be shortened.

15.6.4 A fall in the price of backstop technology

In the model developed in this chapter, we have assumed there is a choke price, *K*. If the net price were to rise above *K*, the economy will cease consumption of the non-renewable resource and switch to an alternative source – the backstop source.

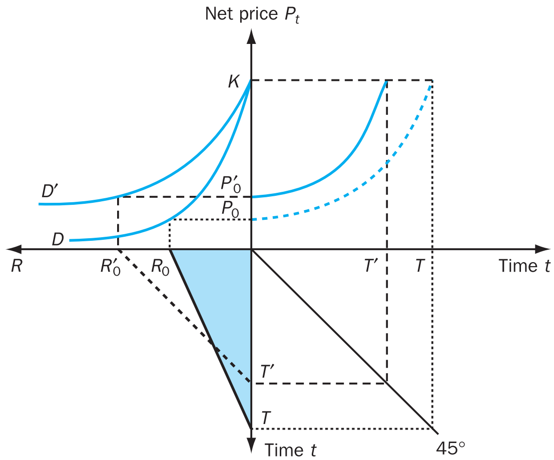


Figure 15.9 The effect of an increase in the demand for the resource

Suppose that technological progress occurs, increasing the efficiency of a backstop technology. This will tend to reduce the price of the backstop source, to P_B ($P_B < K$). Hence the choke price will fall to P_B. Given the fall in the choke price to P_B, the initial value of the resource net price on the original optimal price path, P_0, cannot now be optimal. In fact, it is too high since the net price would reach the new choke price before *T*, leaving some of the economically useful resource unexploited. So the initial price of the non-renewable resource, P_0, must fall to a lower level, P'_0, to encourage an increase in demand so that a shorter time horizon is required until complete exhaustion of the non-renewable resource reserve.

This process is illustrated in Figures 15.10(a) and (b).[14] To correctly interpret what happens when there is a fall in the price of a backstop technology, we need to be a little careful in the way these diagrams are interpreted. Note first that for both scenarios being discussed in this example – with the initial, high choke price, and with the second, lower choke price, a rational plan will draw down the remaining stock to zero at exactly the point in time where the net price reaches the choke price. This does not, of course, mean that the extraction rate, *R*, has to be near to zero at points in time just

[14] I am very grateful to Krister Hjalte for alerting me to a mistake in Figure 15.10 and in the discussion surrounding it in the third edition of this text.

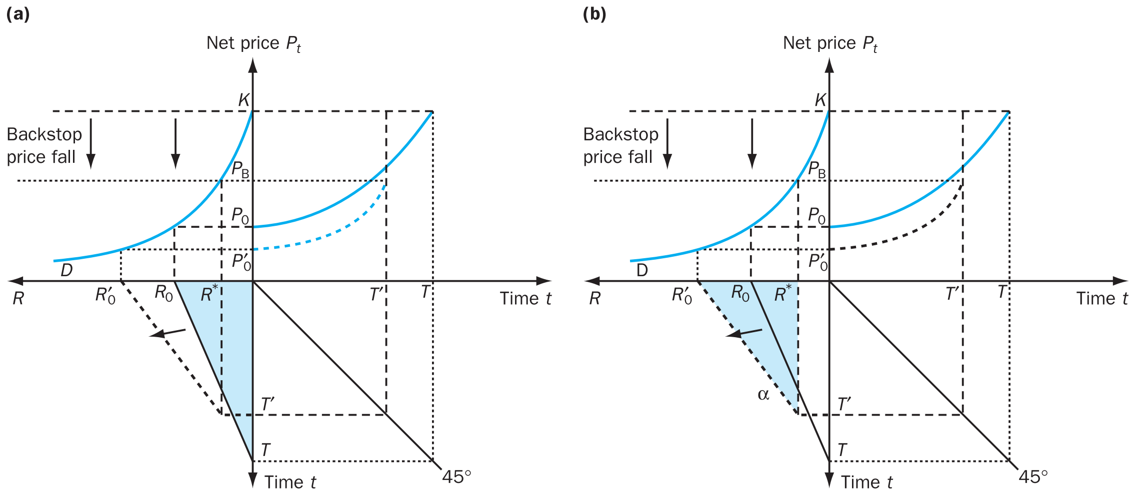


Figure 15.10 (a) A fall in the price of a backstop technology: initial high choke price; (b) A fall in the price of a backstop technology: final low choke price

prior to the switch to the backstop. But nevertheless, the total amount extracted must be identical in the two cases.

The dotted line corresponding to the extraction quantity over time to the time of exhaustion under the new lower choke price scenario needs to have the following properties. First, it will be a negatively sloping line from $t = 0$ onwards until $t = T'$. Also, it will lie below the continuous, downward-sloping line corresponding to the original, higher, backstop price, as with a lower price the extraction rate will be higher. After $t = T'$ is reached, there is a discontinuity: the extraction rate, R, jumps from the level it was immediately prior to T' (in Figures 15.10(a) and (b), this extraction rate is indicated as R^*) to zero. This is shown as a horizontal section of the dotted line in Figures 15.10(a) and (b). However, this should be interpreted with caution as there is actually a discontinuous jump from a positive extraction rate (R^*) to zero extraction rate at T'.

When correctly drawn there will be two triangles on the diagrams that are the same in area. The first of these (shown in Figure 15.10a) is indicated in light shading. The second is shaded in Figure 15.10b. The latter is formed as the triangle lying between the R axis itself, a vertical line dropped down from the R axis at the resource extraction rate that exists immediately before the choke price is reached (R^*), and the downward-sloping section of the dotted line. These two areas are of equal magnitude as the same resource stock is being depleted in each case.

15.6.5 A change in resource extraction costs

Consider the case of an increase in extraction costs, possibly because labour charges rise in the extraction industry. To analyse the effects of an increase in extraction costs, it is important to distinguish carefully between the net price and the gross price of the resource. Let us define:

$$p_t = P_t - c$$

where p_t is the resource net price, P_t is the gross price of the non-renewable resource, and c is the marginal extraction cost, assumed to be constant. Hotelling's rule requires that the resource *net price* grows at a constant rate, equal to the discount rate (which we take here to be constant at the rate i). Therefore, efficient extraction requires that

$$p_t = p_0 e^{it}$$

Now look at Figure 15.11(a). Suppose that the marginal cost of extraction is at some constant level, c_L, and that the curve labelled *Original net price* describes the optimal path of the net price over time (i.e. it plots $p_t = p_0 e^{it}$); also suppose that the corresponding optimal gross price path is given by the

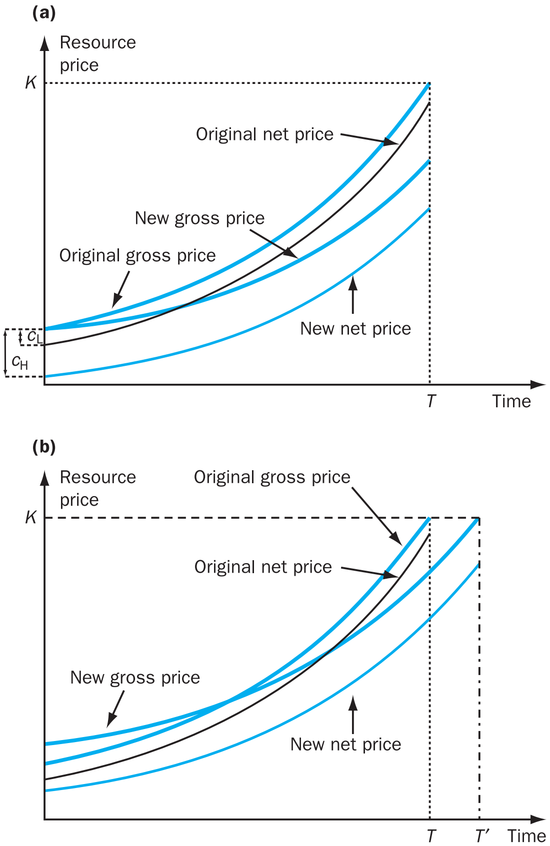


Figure 15.11 (a) An increase in extraction costs: deducing the effects on gross and net prices;
(b) An increase in extraction costs: actual effects on gross and net prices

curve labelled *Original gross price* (i.e. it plots $P_t = p_t + c_L = p_0e^{it} + c_L$).

Next, suppose that the marginal cost of extraction, while still constant, now becomes somewhat higher than was previously the case. Its new level is denoted c_H. We suppose that this change takes place at the initial time period, period 0. Consider first what would happen if the gross price remained unchanged at its initial level, as shown in Figure 15.11(a). The increase in unit extraction costs from c_L to c_H would then result in the net price being lower than its original initial level. However, with no change having occurred in the interest rate, the net price must *grow* at the same rate as before. Although the net price grows at the same rate as before, it does so from a lower starting value, and so it follows that the new net price p_t would be lower at all points in time than the original net price, and it will also have a flatter profile (as close inspection of the diagram makes clear). This implies that the new gross price will be lower than the old gross price at all points in time except in the original period.

However, the positions of the curves for the new gross and net prices in Figure 15.11(a) cannot be optimal. If the gross (market) price is lower at all points in time except period 0, more extraction would take place in every period. This would cause the reserve to become completely exhausted before the choke price (K) is reached. This cannot be optimal, as any optimal extraction path must ensure that demand goes to zero at the same point in time as the remaining resource stock goes to zero.

Therefore, optimal extraction requires that the new level of the gross price in period 0, P'_0, must be greater than it was originally (P_0). It will remain above the original gross price level for a while but will, at some time before the resource stock is fully depleted, fall below the old gross price path. This is the final outcome that we illustrate in Figure 15.11(b). As the new gross price eventually becomes lower than its original level, it must take longer before the choke price is reached. Hence the time taken before complete resource exhaustion occurs is lengthened.

All the elements of this reasoning are assembled together in the four-quadrant diagram shown in Figure 15.12. A rise in extraction costs will raise

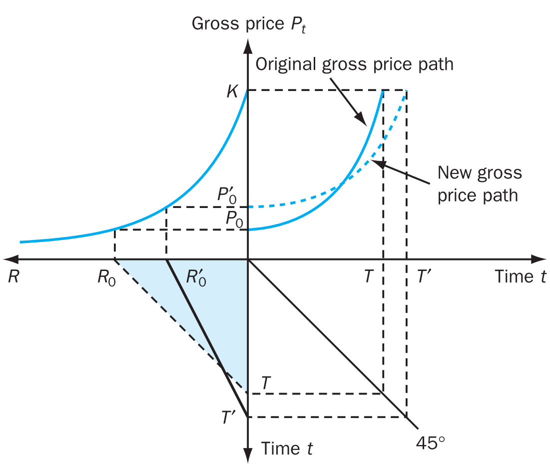


Figure 15.12 A rise in extraction costs

the initial gross price, slow down the rate at which the gross price increases (even though the net price or royalty increases at the same rate as before), and lengthen the time to complete exhaustion of the stock.

What about a fall in extraction costs? This may be the consequence of technological progress decreasing the costs of extracting the resource from its reserves. By following similar reasoning to that we used above, it can be deduced that a fall in extraction costs will have the opposite effects to those just described. It will lower the initial gross price, increase the rate at which the gross price increases (even though the net price increases at the same rate as before), and shorten the time to complete exhaustion of the stock.

If the changes in extraction cost were very large, then our conclusions may need to be amended. For example, if a cost increase were very large, then it is possible that the new gross price in period 0, P'_0, will be above the choke price. It is then not economically viable to deplete the remaining reserve – an example of an economic exhaustion of a resource, even though, in physical terms, the resource stock has not become completely exhausted.

One remaining point needs to be considered. Until now it has been assumed that the resource stock consists of reserves of uniform, homogeneous quality, and the marginal cost of extraction was constant for the whole stock. We have been investigating the consequences of increases or decreases in that marginal cost schedule from one fixed level to another. But what if the stock were not homogeneous, but rather consisted of reserves of varying quality or varying accessibility? It is not possible here to take the reader through the various possibilities that this opens up. It is clear that in this situation marginal extraction costs can no longer be constant, but will vary as different segments of the stock are extracted. There are many meanings that could be attributed to the notion of a change in marginal extraction costs. A fall in extraction costs may occur as the consequence of new, high-quality reserves being discovered. An increase in costs may occur as a consequence of a high-quality mine becoming exhausted, and extraction switching to another mine in which the quality of the resource reserve is somewhat lower. Technical progress may result in the whole profile of extraction costs being shifted downwards, although not necessarily at the same rate for all components.

We do not analyse these cases in this text. The suggestions for further reading point the reader to where analysis of these cases can be found. But it should be evident that elaborating a resource depletion model in any of these ways requires dropping the assumption that there is a known, fixed quantity of the resource. Instead, the amount of the resource that is 'economically' available becomes an endogenous variable, the value of which depends upon resource demand and extraction cost schedules. This also implies that we could analyse a reduction in extraction costs as if it were a form of technological progress; this can increase the stock of the reserve that can be extracted in an economically viable manner. Hence, changes in resource extraction costs and changes in resource stocks become interrelated – rather than independent – phenomena.

15.7 The introduction of taxation/subsidies

15.7.1 A royalty tax or subsidy

A royalty tax or subsidy will have no effect on a resource owner's extraction decision for a reserve that is currently being extracted. The tax or subsidy will alter the present value of the resource being extracted, but there can be no change in the rate of extraction over time that can offset that decline or increase in present value. The government will simply collect some of the mineral rent (or pay some subsidies), and resource extraction and production will proceed in the same manner as before the tax/subsidy was introduced.

This result follows from the Hotelling rule of efficient resource depletion. To see this, define α to be a royalty tax rate (which could be negative – that is, a subsidy), and denote the royalty or net price at time t by p_t. Then the post-tax royalty becomes $(1 - \alpha)p_t$. But Hotelling's rule implies that the post-tax royalty must rise at the discount rate, i, if the resource is to be exploited efficiently. That is:

$$(1 - \alpha)p_t = (1 - \alpha)p_0e^{it}$$

or

$$p_t = p_0e^{it}$$

Hotelling's rule continues to operate unchanged in the presence of a royalty tax, and no change occurs to the optimal depletion path. This is also true for a royalty subsidy scheme. In this case, denoting the royalty subsidy rate by β, we have the efficiency condition

$$(1 + \beta)p_t = (1 + \beta)p_0e^{it} \Rightarrow p_t = p_0e^{it}$$

We can conclude that a royalty tax or subsidy is neutral in its effect on the optimal extraction path. However, a tax may discourage (or a subsidy encourage) the exploration effort for new mineral deposits by reducing (increasing) the expected payoff from discovering the new deposits.

15.7.2 Revenue tax/subsidy

The previous subsection analysed the effect of a tax or subsidy on resource royalties. We now turn our attention to the impact of a revenue tax (or subsidy). In the absence of a revenue tax, the Hotelling efficiency condition is, in terms of net prices and gross prices,

$$p_t = p_0e^{it}$$
$$\Rightarrow (P_t - c) = (P_0 - c)e^{it}$$

Under a revenue tax scheme, with a tax of α per unit of the resource sold, the post-tax royalty or net price is

$$p_t = (1 - \alpha)P_t - c$$

So Hotelling's rule becomes:

$$[(1 - \alpha)P_t - c] = [(1 - \alpha)P_0 - c]e^{it} \quad (0 < \alpha < 1)$$

$$\Rightarrow \left(P_t - \frac{c}{1 + \alpha}\right) = \left(P_0 - \frac{c}{1 + \alpha}\right)e^{it}$$

Since $c/(1 - \alpha) > c$, an imposition of a revenue tax is equivalent to an increase in the resource extraction cost. Similarly, for a revenue subsidy scheme, we have

$$\left(P_t - \frac{c}{1 + \beta}\right) = \left(P_0 - \frac{c}{1 + \beta}\right)e^{it} \quad (0 < \beta < 1)$$

A revenue subsidy is equivalent to a decrease in extraction cost. We have already discussed the effects of a change in extraction costs, and you may recall the results we obtained: a decrease in extraction costs will lower the initial gross price, increase the rate at which the gross price increases (even though the net price or royalty increases at the same rate as before) and shorten the time to complete exhaustion of the stock.

15.8 The resource depletion model: some extensions and further issues

15.8.1 Discount rate

We showed above that resource extraction under a system of perfectly competitive markets might produce the socially optimal outcome. But this equivalence rests upon several assumptions, one of which is that firms choose a private discount rate identical to the social discount rate that would be used by a rational planner. If private and social discount rates differ, however, then market extraction paths may be biased toward excessive use or conservation relative to what is socially optimal.

15.8.2 Forward markets and expectations

The Hotelling model is an abstract analytical tool; its operation in actual market economies is dependent upon the existence of a set of particular institutional circumstances. In many real situations these institutional arrangements do not exist and so the rule lies at a considerable distance from the operation of actual market mechanisms. In addition to the discount rate equivalence mentioned in the previous section, two assumptions are required to ensure a social optimal extraction in the case of perfect competition. First, the resource must be owned by the competitive agents. Secondly, each agent must know at each point in time all current and future prices. One might just assume that agents have perfect foresight, but this hardly seems tenable for the case we are investigating. In the absence of perfect foresight, knowledge of these prices requires the existence of both spot markets and a complete set of forward markets for the resource in question. But no resource does possess a complete set of forward markets, and in these circumstances there is no guarantee that agents can or will make rational supply decisions.

15.8.3 Optimal extraction under uncertainty

Uncertainty is prevalent in decision making regarding non-renewable resource extraction and use. There is uncertainty, for example, about stock sizes, extraction costs, how successful research and development will be in the discovery of substitutes for non-renewable resources (thereby affecting the cost and expected date of arrival of a backstop technology), pay-offs from exploration for new stock, and the action of rivals. It is very important to study how the presence of uncertainty affects appropriate courses of action. For example, what do optimal extraction programmes look like when there is uncertainty, and how do they compare with programmes developed under conditions of certainty?

Let us assume an owner of a natural resource (such as a mine) wishes to maximise the net present value of utility over two periods:[15]

$$\text{Max}\left(U_0 + \frac{U_1}{1+\rho}\right)$$

If there is a probability (π) of a disaster (for example, the market might be lost) associated with the second period of the extraction programme, then the owner will try to maximise the *expected* net present value of the utility (if he or she is risk-neutral):

$$\text{Max}\left(U_0 + \pi \cdot 0 + (1-\pi)\frac{U_1}{1+\rho}\right)$$

$$= \text{Max}\left(U_0 + (1-\pi)\frac{U_1}{1+\rho}\right) = \text{Max}\left(U_0 + \frac{U_1}{1+\rho^*}\right)$$

where

$$\frac{1}{1+\rho^*} = \frac{1-\pi}{1+\rho}$$

Note that

$$(1+\rho^*)(1-\pi) = 1+\rho$$

$$\Rightarrow \rho^* - \rho = \pi(1+\rho^*) > 0 \qquad (\text{if } 1 \geq \pi > 0)$$

$$\Rightarrow \rho^* > \rho$$

Therefore, in this example, the existence of risk is equivalent to an increase in the discount rate for the owner, which implies, as we have shown before, that the price of the resource must rise more rapidly and the depletion is accelerated.

15.9 Do resource prices actually follow the Hotelling rule?

The Hotelling rule is an economic theory. It is a statement of how resource prices should behave under a specified (and very restrictive) set of conditions. Economic theory begins with a set of axioms (which are regarded as not needing verification) and/or a set of assumptions (which are treated as being provisionally correct). These axioms or assumptions typically include goals or objectives of the relevant actors and various rules of how those actors behave. Then logical reasoning is used to deduce outcomes that should follow, given those assumptions.

But a theory is not necessarily correct. Among the reasons it may be wrong are inappropriateness of one or more of its assumptions, and flawed deduction. A theory may also fail to 'fit the facts' because it refers to an idealised model of reality that does not take into account some elements of real-world complexity. However, failing to fit the facts does not make the theory *false*; the theory only applies exactly to the idealised construct that it was intended to analyse.

But it would be interesting to know whether the Hotelling principle is sufficiently powerful to fit the facts of the real world. Indeed, many economists take the view that a theory is useless unless it has predictive power: we should be able to use the theory to make predictions that have a better chance of being correct than chance alone would imply. A theory is unlikely to have predictive power if

[15] This argument follows very closely a presentation in Fisher (1981).

it cannot describe or explain current and previous behaviour. Of course, even if it could do that, this does not necessarily mean it will have good *ex ante* predictive power.

In an attempt to validate the Hotelling rule (and other associated parts of resource depletion theory), much research effort has been directed to empirical testing of that theory. What conclusions have emerged from this exercise? Unfortunately, no consensus of opinion has come from empirical analysis. As Berck (1995) writes in one survey of results 'the results from such testing are mixed'.

A simple version of the Hotelling rule for some marketed non-renewable resource was given by equation 15.7b; namely

$$\frac{\dot{p}_t}{p_t} = \rho$$

In this version, all prices are denominated in units of utility, and ρ is a utility discount rate. These magnitudes are, of course, unobservable, so equation 15.7b is not directly testable. But we can rewrite the Hotelling rule in terms of money-income (or consumption) units that can be measured:

$$\frac{\dot{p}^*_t}{p^*_t} = r \tag{15.10}$$

Here, p^* denotes a price in money units, and r is a consumption discount rate. Empirical testing normally uses discrete time-series data, and so the discrete-time version of Hotelling's rule is employed:

$$\frac{\Delta p^*_t}{p^*_t} = r \tag{15.11}$$

or, expressed in an alternative way,

$$p^*_{t+1} = p^*_t(1 + r) \tag{15.12}$$

Notice right away that equations 15.11 and 15.12 are assuming that there is a constant discount rate over time. If this is not correct (and there is no reason why it has to be) then r should enter those two equations with a time subscript, and the Hotelling principle no longer implies that a resource price will rise at a fixed rate. But this is a complication we ignore in the rest of this section.

One way of testing Hotelling's rule seems to be clear: collect time-series data on the price of a resource, and see if the proportionate growth rate of the price is equal to δ. This was one thing that Barnett and Morse (1963) did in a famous study. They found that resource prices – including iron, copper, silver and timber – fell over time, which was a most disconcerting result for proponents of the standard theory. Subsequent researchers, looking at different resources or different time periods, have come up with a bewildering variety of results. There is no clear picture of whether resource prices typically rise or fall over time. We can no more be confident that the theory is true than that it is not true – a most unsatisfactory state of affairs.

But we now know that the problem is far more difficult than this to settle, and that a direct examination of resource prices is not a reasonable way to proceed. Note first that the variable p^* in Hotelling's rule is the *net* price (or rent, or royalty) of the resource, not its *market* price. Roughly speaking, these are related as follows:

$$P^* = p^* + \text{MC} \tag{15.13}$$

where P^* is the gross (or market) price of the extracted resource, p^* is the net price of the resource *in situ* (i.e. unextracted), and MC is the marginal extraction cost. It is clear from equation 15.13 that if the marginal cost of extraction is falling, P^* might be falling even though p^* is rising. We noted this earlier in doing comparative statics to examine the effect of a fall in extraction costs. So evidence of falling market prices cannot, in itself, be regarded as invalidating the Hotelling principle.

This suggests that the right data to use is the resource net price. But that is an unobservable variable, for which data do not therefore exist. And this is not the only unobservable variable: δ is also un-observed, as we shall see shortly. In the absence of data on net price, one might try to construct a proxy for it. The obvious way to proceed is to subtract marginal costs from the gross, market price to arrive at net price. This is also not as easy as it seems: costs are observable, but the costs recorded are usually averages, not marginals. We shall not discuss how this (rather serious) difficulty has been dealt with. However, many studies have pursued this approach. Slade (1982) made one of the earliest studies of this type; she concluded that some resources have U-shaped quadratic price paths, having fallen in the past but latterly rising. Other studies of this type are Stollery (1983), which generally supported the Hotelling hypothesis, and Halvorsen and Smith (1991), which was unable to support it.

Any attempt to construct a proxy measure for net price comes up against an additional problem. The measure that is obtained is a proxy, and it will contain estimation errors. If this variable is simply treated as if it were the unobserved net price itself, a statistical problem – known to econometricians as an errors-in-variables problem – will occur, and estimates of parameters will in general be biased (and so misleading) no matter how large is the sample of data available to the researcher. This casts doubt on all studies using proxies for the net price which have not taken account of this difficulty. Appropriate statistical techniques in the presence of errors-in-variables are discussed in most intermediate econometrics texts, such as Greene (1993). Harvey (1989) is a classic text on the Kalman filter, which is one way of resolving this problem.

Other approaches have also been used to test the Hotelling rule, and we shall mention only two of them very briefly. Fuller details can be found in the survey paper by Berck (1995). Miller and Upton (1985) use the valuation principle. This states that the stock market value of a property with unextracted resources is equal to the present value of its resource extraction plan; if the Hotelling rule is valid this will be constant over time, and so the property's stock market value will be constant. Evidence from this approach gives reasonably strong support for the Hotelling principle. Farrow (1985) adopts an approach that interprets the Hotelling rule as an asset-efficiency condition, and tests for efficiency in resource prices, in much the same way that finance theorists conduct tests of market efficiency. These tests generally reject efficiency, and by implication are taken to not support the Hotelling rule. However, it has to be said that evidence in favour of efficient asset markets is rarely found, but that does not stop economists assuming (for much of the time) that asset markets are efficient.

Let us now return to a comment we made earlier. The right-hand side of the Hotelling rule equation consists of the consumption discount rate δ. But this is also a theoretical construct, not directly observable. What we do observe are market rates of interest, which will include components reflecting transaction costs, various degrees of risk premia, and other market imperfections. Even if we could filter these out, the market rate of interest measures realised or *ex post* returns; but the Hotelling theory is based around an *ex ante* measure of the discount rate, reflecting expectations about the future. This raises a whole host of problems concerning how expectations might be proxied that are beyond the scope of this text.

Finally, even if we did find convincing evidence that the net price of a resource does not rise at the rate δ (or even that it falls), should we regard this as evidence that invalidates the Hotelling rule? The answer is that we should not draw this conclusion. There are several circumstances where resource prices may fall over time even where a Hotelling rule is being followed. For example, in Figure 15.8 we showed that a sequence of new mineral discoveries could lead to a downward-sloping path of the resource's net price. Pindyck (1978) first demonstrated this in a seminal paper. If resource extraction takes place in non-competitive markets, the net price will also rise less quickly than the discount rate (see Figure 15.4). And in the presence of technical progress continually reducing extraction costs, the market price may well fall over time, thereby apparently contradicting a simple Hotelling rule.

The history of attempts to test the Hotelling principle is an excellent example of the problems faced by economists in all branches of that discipline. Many of the variables used in our theories are unobservable or latent variables. Shadow prices are one class of such latent variables. The best we can do is to find proxy variables for them. But if the theory does not work, is that because the theory is poor or because our proxy was not good? More generally, a theory pertains to a particular model. So unless it contains a logical error, a theory can never be wrong. What can be, and often is, incorrect, is a presumption that a theory that is correct in the context of one particular model will generate conclusions that are valid in a wide variety of 'real' situations.

15.10 Natural resource scarcity

Concern with the supposed increasing scarcity of natural resources, and the possibility of running out of strategically important raw materials or energy sources, is by no means new. Worries about resource scarcity can be traced back to medieval times in Britain, and have surfaced periodically ever since. The scarcity of land was central to the theories of Malthus and the other classical economists. In the 20th century, fears about timber shortages in several countries led to the establishment of national forestry authorities, charged with rebuilding timber stocks. As we have seen earlier, pessimistic views about impending resource scarcity have been most forcibly expressed in the *Limits to Growth* literature (see Chapter 2 of this text for examples); during the 1970s, the so-called oil crises further focused attention on mineral scarcities.

What do we mean by resource scarcity? One use of the term – what might be called absolute scarcity – holds that all resources are scarce, as the availability of resources is fixed and finite at any point in time, while the wants which resource use can satisfy are not limited. Where a market exists for a resource, the existence of any positive price is viewed as evidence of absolute scarcity; where markets do not exist, the existence of a positive shadow price – the implicit price that would be necessary if the resource were to be used efficiently – similarly is an indicator of absolute scarcity for that resource.

But this is not the usual meaning of the term in general discussions about natural resource scarcity. In these cases, scarcity tends to be used to indicate that the natural resource is becoming harder to obtain, and requires more of other resources to obtain it. The relevant costs to include in measures of scarcity are both private and external costs; it is important to recognise that if private extraction costs are not rising over time, social costs may rise if negative externalities such as environmental degradation or depletion of common property resources are increasing as a consequence of extraction of the natural resource. Thus, a rising opportunity cost of obtaining the resource is an indicator of scarcity – let us call this use of the term *relative scarcity*. In the rest of this section, our comments will be restricted to this second form.

Before we take this matter any further, it is necessary to say something about the degree of aggregation used in examining resource scarcity. To keep things as simple as possible, first consider only non-renewable natural resources. There is not one single resource but a large number, each distinct from the others in some physical sense. However, physically distinct resources may be economically similar, through being substitutes for one another. Non-renewable resources are best viewed, then, as a structure of assets, components of which are substitutable to varying degrees. In Chapter 14, when we discussed the efficient extraction of a single non-renewable resource, what we had in mind was some aggregate set of resources in this particular sense. Moreover, when the class of resources is extended to incorporate renewable resources, so the structure is enlarged, as are the substitution possibilities.

Except for resources for which no substitution possibilities exist – if indeed such resources exist – it is of limited usefulness to enquire whether any individual resource is scarce or not. If one particular resource, such as crude oil, were to become excessively costly to obtain for any reason, one would expect resource use to substitute to another resource, such as natural gas or coal. A well-functioning price mechanism should ensure that this occurs. Because of this, it is more useful to consider whether natural resources in general are becoming scarcer: is there any evidence of increasing generalised resource scarcity?

What indicators might one use to assess the degree of scarcity of particular natural resources, and natural resources in general? There are several candidates for this task, including physical indicators (such as reserve quantities or reserve-to-consumption ratios), marginal resource extraction cost, marginal exploration and discovery costs, market prices, and resource rents. We shall now briefly examine each of these. In doing so, you will see that the question of whether resources are becoming scarce is closely related to the question of whether the Hotelling rule is empirically validated.

15.10.1 Physical indicators

A variety of physical indicators have been used as proxies for scarcity, including various measures of reserve quantities, and reserve-to-consumption ratios. Several such measures were discussed earlier in this chapter and appropriate statistics listed (see Box 15.1 and Table 15.1). Inferences drawn about impending resource scarcity in the *Limits to Growth* literature were drawn on the basis of such physical indicators. Unfortunately, they are severely limited in their usefulness as proxy measures of scarcity for the reasons discussed in Box 15.1. Most importantly, most natural resources are not homogeneous in quality, and the location and quantities available are not known with certainty; extra amounts of the resource can be obtained as additional exploration, discovery and extraction effort is applied. A rising resource net price will, in general, stimulate such effort. It is the absence of this information in physical data that limits its usefulness.

15.10.2 Real marginal resource extraction cost

We argued earlier that scarcity is concerned with the real opportunity cost of acquiring additional quantities of the resource. This suggests that the marginal extraction cost of obtaining the resource from existing reserves would be an appropriate indicator of scarcity. The classic study by Barnett and Morse (1963) used an index of real unit costs, c, defined as

$$c = \frac{(\alpha L + \beta K)}{Q}$$

where L is labour, K is capital and Q is output of the extractive industry, and α and β are weights to aggregate inputs. Rising resource scarcity is proxied by rising real unit costs. Note that ideally marginal costs should be used, although this is rarely possible in practice because of data limitations. An important advantage of an extraction costs indicator is that it incorporates technological change. If technological progress relaxes resource constraints by making a given quantity of resources more productive, then this reduction in scarcity will be reflected in a tendency for costs to fall. However, the measure does have problems. First, the measurement of capital is always difficult, largely because of the aggregation that is required to obtain a single measure of the capital stock. Similarly, there are difficulties in obtaining valid aggregates of all inputs used. Secondly, the indicator is backward-looking, whereas an ideal indicator should serve as a signal for future potential scarcity. Finally, it may well be the case that quantities and/or qualities of the resource are declining seriously, while technical progress that is sufficiently rapid results in price falling. In extreme cases, sudden exhaustion may occur after a period of prolonged price falls. Ultimately, no clear inference about scarcity can be drawn from extraction cost data alone.

Barnett and Morse (1963) and Barnett (1979) found no evidence of increasing scarcity, except for forestry. As we mentioned previously, they concluded that agricultural and mineral products, over the period 1870 to 1970, were becoming more abundant rather than scarcer, and explained this in terms of the substitution of more plentiful lower-grade deposits as higher grades were depleted, the discovery of new deposits, and technical change in exploration, extraction and processing. References for other, subsequent studies are given at the end of the chapter.

15.10.3 Marginal exploration and discovery costs

An alternative measuring of resource scarcity is the opportunity cost of acquiring additional quantities of the resource by locating as-yet-unknown reserves. Higher discovery costs are interpreted as indicators of increased resource scarcity. This measure is not often used, largely because it is difficult to obtain long runs of reliable data. Moreover, the same kinds of limitations possessed by extraction cost data apply in this case too.

15.10.4 Real market price indicators and net price indicators

The most commonly used scarcity indicator is time-series data on real (that is, inflation-adjusted) market prices. It is here that the affinity between tests of scarcity and tests of the Hotelling principle is most apparent. Market price data are readily available, easy to use and, like all asset prices, are forward-looking, to some extent at least. Use of price data has three main problems. First, prices are often distorted as a consequence of taxes, subsidies, exchange controls and other governmental interventions; reliable measures need to be corrected for such distortions. Secondly, the real price index tends to be very sensitive to the choice of deflator. Should nominal prices be deflated by a retail or wholesale price index (and for which basket of goods), by the GDP deflator, or by some input price index such as manufacturing wages? There is no unambiguously correct answer to this question, which is unfortunate as very different conclusions can be arrived at about resource scarcity with different choices of deflator. Some evidence on this is given in the chapter on resource scarcity in Hartwick and Olewiler (1986); these authors cite an analysis by Brown and Field (1978) which compares two studies of resource prices using alternative deflators. For eleven commodities, Nordhaus (1973) used capital goods prices as a deflator and concluded that all eleven minerals were becoming less scarce. However, Jorgensen and Griliches (1967) used a manufacturing wages deflator and concluded that three of the minerals – coal, lead and zinc – were becoming scarcer over the same period.

The third major problem with resource price data is one we came across earlier. Market prices do not in general measuring the right thing; an ideal price measure would reflect the net price of the resource. Hotelling's rule shows that it is this that rises through time as the resource becomes progressively scarcer. But we have seen that net resource prices are not directly observed variables, and so it is rather difficult to use them as a basis for empirical analysis.

Despite the limitations of market price data, the early studies show a broad agreement between this measure and the others discussed in this section. One illustration is given in Figure 15.13, taken from Brown and Field (1979), which suggests that, for an aggregate index of all metals, scarcity was decreasing over the period 1890 to 1970. More recent studies present a much less clear picture, however – as we noted above.

Can any general conclusions about resource scarcity be obtained from the literature? The majority of economic analyses conducted up to the early 1980s concluded that few, if any, non-renewable natural resources were becoming scarcer. In the last 20 years, concern about increasing scarcity of non-renewable resources has increased, and an increasing proportion of studies seems to lend support to an increasing scarcity hypothesis.

Paradoxically, these studies also suggested it was in the area of *renewable* resources that problems of increasing scarcity were to be found, particularly in cases of open access. The reasons why scarcity may be particularly serious for some renewable resources will be examined in Chapter 17.

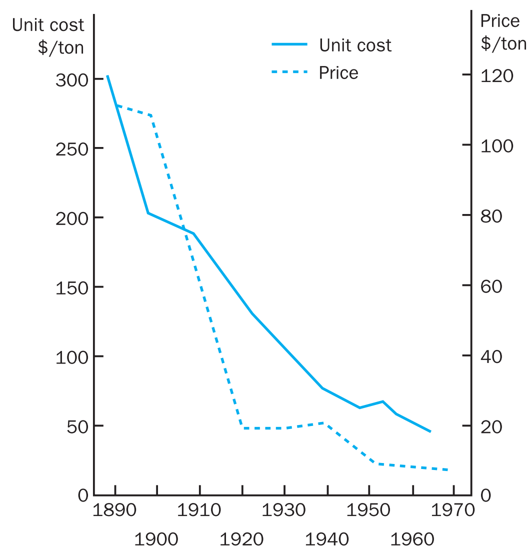


Figure 15.13 Price and unit costs for all metals, 1890–1970
Source: Brown and Field (1979). Copyright, Resources for the Future, Inc.

Summary

- Non-renewable resources consist of energy and material stocks that are generated very slowly through natural processes; these stocks – measured in terms of base resource – can be thought of as existing in fixed, finite quantities. Once extracted, they cannot regenerate in timescales that are relevant to humans.

- Resource stocks can be measured in several ways, including base resource, resource potential, and resource reserves. It is important to distinguish between purely physical measures of stock size, and 'economic' measures of resource stocks.
- Non-renewable resources consist of a large number of particular types and forms of resource, among which there may be substitution possibilities.
- The demand for a resource may exhibit a 'choke price'; at such a price demand would become zero, and would switch to an alternative resource or to a 'backstop' technology.
- The chapter has shown – for two-period discrete time and for continuous time – how models of optimal resource depletion can be constructed and solved.
- One element of the solution of such models is that an efficient price path for the non-renewable resource must follow the Hotelling rule.
- In some circumstances, a socially optimal depletion programme will be identical to a privately optimal (profit-maximising) depletion programme. However, this is not always true. In particular, the equivalence will not hold if social and private discount rates diverge.
- Using comparative dynamic analysis, we have been able to determine the consequences of changes in interest rates, known stock size, demand, price of backstop technology, and resource extraction costs.
- Frequent new discoveries of the resource are likely to generate a price path which does not resemble constant exponential growth.
- Resource depletion outcomes differ between competitive and monopolistic markets. The time to depletion will be longer in a monopoly market, the resource net price will be higher in early years, and the net price will be lower in later years.
- Taxes or subsidies on royalties (or resource rents or net prices) will not affect the optimal depletion path, although they will affect the present value of after-tax royalties. However, revenue-based taxes or subsidies will affect depletion paths, being equivalent to changes in extraction costs.
- We explained the concept of natural resource scarcity. There are many measures that have been proposed, or are used, as measures of scarcity. The more theoretically attractive measures typically are unobtainable as they depend upon unobservable quantities.

Further reading

The references for further reading given at the end of Chapter 14 are all relevant for further reading on the material covered in this chapter. In particular, very good (but rather advanced-level) presentations of the theory of efficient and optimal resource depletion can be found in Baumol and Oates (1988), Dasgupta and Heal (1979), Heal (1981) and the collection of papers in the May 1974 special issue on resource depletion of the *Review of Economic Studies*. As stated previously, less difficult presentations are given in Hartwick and Olewiler (1986), Anderson (1991) and Fisher (1981). Pindyck (1978) is the classic reference on resource exploration.

Good general discussions of resource scarcity can be found in Hartwick and Olewiler (1986, chapter 5), which provides an extensive discussion of the evidence, Barbier (1989a), Fisher (1979, 1981) and Harris (1993). Important works in the field of resource scarcity include Barnett (1979), Barnett and Morse (1963), Brown and Field (1979), Deverajan and Fisher (1980, 1982), Hall and Hall (1984), Jorgensen and Griliches (1967), Leontief *et al.* (1977), Nordhaus (1973), Norgaard (1975), Slade (1982), Smith (1979) and Smith and Krutilla (1979). Examinations of the extent to which the Hotelling rule are satisfied in practice are extensively referenced in the text, but the best place to go next is probably Berck (1995).

An excellent discussion on natural resource substitutability can be found in Dasgupta (1993). Adelman (1990, 1995) covers the economics of oil depletion. Prell (1996) deals with backstop technology.

Discussion questions

1. Discuss the merits of a proposal that the government should impose a tax or subsidy where an non-renewable resource is supplied monopolistically in order to increase the social net benefit.

2. 'An examination of natural resource matters ought to recognise technical/scientific, economic, and socio-political considerations.' Explain.

3. 'The exploitation of resources is not necessarily destructive . . . need not imply the impoverishment of posterity . . . It is the diversion of national income from its usual channels to an increased preservation of natural wealth that will harm posterity' (Anthony Scott). Explain and discuss.

4. The notion of sustainability is used differently in economics than in the natural sciences. Explain the meaning of sustainability in these two frameworks, and discuss the attempts that have been made by economists to make the concept operational.

Problems

1. Consider two consecutive years, labelled 0 and 1. You are currently at the start of year 0. The following information is available. There is a single fixed stock of a non-renewable resource; the magnitude of this stock at the start of year 0 is 224 (million tonnes). The inverse resource demand functions for this resource in each of the years are

$$P_0 = a - bR_0 \quad \text{and} \quad P_1 = a - bR_1$$

in which $a = 107$ and $b = 1$. The constant marginal cost of resource extraction is 5. All (non-physical) units are in European units of utility. The social welfare function is discounted utilitarian in form, with a social utility discount rate of 0.1. Given that the objective is to maximise social welfare over periods 0 and 1, calculate the amounts of resource that should be extracted in each period, subject to the restriction that at least 104 units of the resource should be left (unextracted) for the future at the end of period 1. What is the resource price in each period
(a) in utility units;
(b) in euros, given that $U = \log(C)$, where U is utility units, log is the natural logarithm operator, and C is consumption (or income), measured in euros?

2. The version of Hotelling's rule given in equation 15.5 requires the net price to grow proportionately at the rate ρ. Under what circumstances would this imply that the gross price also should grow at the rate ρ?

3. In equation 15.5, if $\rho = 0$, what are the implications for
(a) P_0 and P_1?
(b) R_0 and R_1?

(Problems 4, 5 and 6 are based on Table 15.3.)

4. Explain, with diagrams, why a monopolistic non-renewable resource market is biased towards conservation and therefore will increase the 'life' of the resource.

5. In the case of perfect competition, if the private discount rate is higher than the correct social discount rate, explain, with diagrams, why the market will exhaust the resource too quickly.

6. Discuss, with diagrams, the consequences of the discovery of North Sea oil for
(a) the price and output levels for the oil market;
(b) the date of exhaustion of oil reserves.

What will be the probable path over time of oil prices if there are frequent discoveries of oil?

CHAPTER 16 Stock pollution problems

Look before you leap.

Proverb, source unknown; most likely source is Aesop's fables

Learning objectives

In this chapter you will

- investigate two models of optimal emissions which are suitable for the analysis of persistent (long-lasting) pollutants. Each of these models is a variant of the optimal growth model framework that we have addressed before at several places in the text
- investigate a simple 'aggregate stock pollution model'. This model is appropriate for dealing with pollution problems where the researcher considers it appropriate to link emissions flows to the processes of resource extraction and use
- use the aggregate stock pollution model to identify how optimal pollution targets can be obtained from generalised versions of the resource depletion models we investigated in Chapters 14 and 15
- follow the development of a second resource use and depletion model. This model – which we call a 'model of waste accumulation and disposal' – provides a framework that is suitable for analysing stock pollution problems of a local, or less pervasive, type, such as the accumulation of lead in water systems or contamination of water systems by effluent discharges
- investigate in some depth the dynamics of pollution generation and pollution regulation processes, using phase plane analysis

Introduction

Our analysis of pollution targets in Chapter 5 recognised that some residuals are durable. Their emissions accumulate, impose loads upon environmental systems which persist through time, and can result in harmful impacts. Processes of this form were called stock pollution problems. In this chapter, we revisit our previous analysis of pollution targets (in Chapter 5). Two modelling frameworks will be examined. We refer to these as an 'aggregate stock pollution model' and a 'model of waste accumulation and disposal'.[1] The first is appropriate for dealing with pollution problems at a highly aggregated level, and where it is necessary to place pollution problems explicitly in the context of the material basis of the economy, by linking residual flows to the processes of resource extraction and use. In doing so, it will be possible to generate pollution targets from the resource depletion models we investigated in Chapters 14 and 15.

This approach is appropriate for dealing with economy-wide or global stock pollution problems arising from the use of fossil fuels. Climate change modelling falls into this category, and several of the illustrations we use in the chapter refer to that example. Most climate change models are highly

[1] The term 'model of waste accumulation and disposal' is borrowed from the title of Plourde's (1972) seminal paper.

aggregated using, for example, an aggregate 'fossil fuels' resource as an input into production. And they require that the material basis of the pollution in question – in this case, finite stocks of fossil fuels – is properly built into the modelling framework.

The second framework – the waste accumulation and disposal model – is appropriate for analysing stock pollution problems of a local, or less pervasive, type. Examples of such problems include the accumulation of lead, mercury and other heavy metals in water systems, the accumulation of particulates in air, the build-up of chemicals from pesticides and fertilisers in soils, and contamination of inland and coastal water systems by effluent discharges. In these cases, resource use is of a sufficiently small scale (in the problem being considered) that limits on resource stocks do not become binding constraints. Hence, the researcher can focus on the dynamics of the pollution problem but need not explicitly build into the model a component which links pollutant emissions to the resources from which they are derived.

For both modelling frameworks, though, we shall take the analysis of previous chapters further by giving a more complete account of the dynamics of the pollution processes, the properties of their steady states (if they exist), and the implications for pollution control targets and instruments.

16.1 An aggregate dynamic model of pollution

Pollution problems come in many forms. Yet many have one thing in common: they are associated with the use of fossil fuels. In this section, we present a simple and highly aggregated stock pollution model. To fix ideas, it will be useful to think of this as a global climate change model, although that is by no means the only context in which the model could be used.

16.1.1 Basic structure of the model

The model developed in this section is a simple, aggregate stock pollution model. It can be thought of as an optimal growth model – of the type covered in Chapter 14 – but including some additional components, one of which models the way in which pollution flows are related to the extraction and use of a composite non-renewable resource. We employ here equivalent notation to that used in Chapter 14 and, wherever appropriate, adopt equivalent functional forms. Being an optimal growth model, we look for its 'solution' by using dynamic optimisation techniques. Specifically, we are trying to find the characteristics of an emissions path for the pollutant that will maximise a suitably defined objective function.

We suppose that the production process utilises two inputs: capital and a non-renewable environmental resource. Obtaining that non-renewable resource involves extraction and processing costs. There is a fixed (and known) total stock of the non-renewable resource. From now on we shall refer to this resource as 'fossil fuels'. Use of fossil fuels involves two kinds of trade-offs. First, there is an intertemporal trade-off: given that the total stock is fixed, using fossil fuels today means that less will be available tomorrow. So different paths of fossil-fuel extraction can affect the welfare of different generations. Second, using fossil fuels leads to more production (which is welfare-enhancing) but also generates more pollution (which is welfare-reducing). The principal concern of Chapters 14 and 15 was with the intertemporal trade-off. Here we are interested in both of these trade-offs.

The pollution model used is an extension of that developed in Chapter 14. Its structure – elements and key relationships – is illustrated in Figure 16.1. We retain the assumption that extracting the resource is costly, but simplify the earlier analysis by having those costs dependent on the rate of extraction but not on the size of the remaining stock. Pollution is generated from the use of the fossil-fuel resource.

16.1.2 Pollution damages

There are various ways in which pollution damages can be incorporated into a resource depletion model. Two of these are commonly used in environmental economics:

- damages operating through the utility function;
- damages operating through the production function.

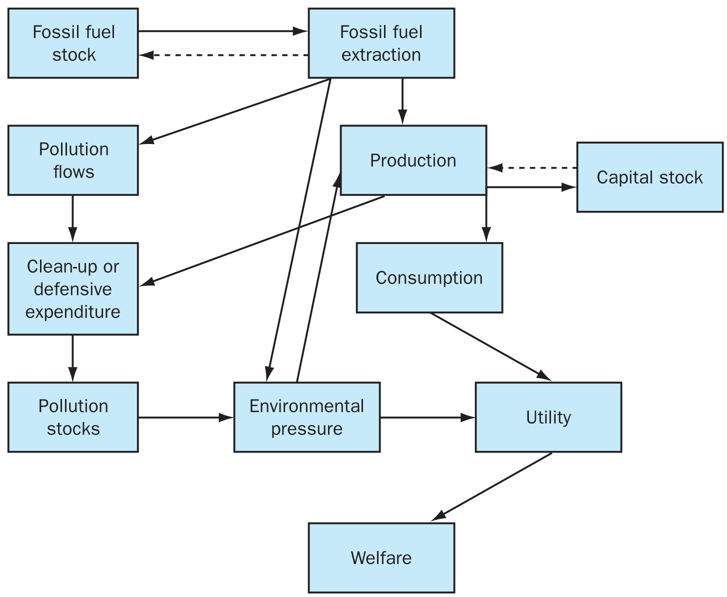


Figure 16.1 The structure of the aggregate stock pollution model

In order to handle these kinds of effects in a fairly general way, we use the symbol E to denote an index of environmental pressures. These environmental pressures have a negative effect upon utility. To capture these effects, we write the utility function as

$$U = U(C, E) \tag{16.1}$$

in which, by assumption, $U_C > 0$ and $U_E < 0$. The index of environmental pressures E depends on the rate of fossil-fuel use (R) and on the accumulated stock of pollutant in the relevant environmental medium (A). So we have

$$E = E(R, A) \tag{16.2}$$

Higher rates of fossil-fuel use and higher ambient pollution levels each increase environmental pressures, so that $E_R > 0$ and $E_A > 0$. Substituting equation 16.2 into equation 16.1 we obtain

$$U = U(C, E(R, A)) \tag{16.3}$$

This deals with the case where damages operate through the utility function. But many forms of damage operate through production functions. For example, greenhouse-gas-induced climate change might reduce crop yields, or tree growth may be damaged by sulphur dioxide emissions. A production function that incorporates damages of this kind is

$$Q = Q(R, K, E(R, A)) \tag{16.4}$$

Obtaining the non-renewable resource involves extraction and processing costs, Γ, which depend on the quantity of the resource used; hence we have

$$\Gamma = \Gamma(R)$$

16.1.3 The resource stock–flow relationship

The utility and production functions both depend on A, the ambient level of pollution. The way in which A changes over time is modelled in the same way as in Chapter 5. That is:

$$\dot{A} = M(R) - \alpha A \tag{16.5}$$

which assumes that a constant proportion α of the ambient pollutant stock decays at each point in time. Note that equation 16.5 specifies that emissions depend upon the amount of resource use, R. By integration of equation 16.5 we obtain

$$A_t = \int_{\tau=0}^{t} (M(R_\tau) - \alpha A_\tau)\mathrm{d}\tau$$

So for a pollutant which is not infinitely long-lived ($\alpha > 0$) the pollution stock at time t will be the sum of all previous pollution emissions less the sum of all previous pollution decay, while for a perfectly persistent pollutant ($\alpha = 0$) A grows without bounds as long as M is positive.

16.1.4 Clean-up expenditure

We now introduce an additional control variable (or instrument) – expenditure on cleaning-up pollution, V. Expenditure on V is an alternative use of output to investment expenditure, consumption, or resource extraction and processing costs, and so must satisfy the identity

$$Q \equiv \dot{K} + C + \Gamma + V$$

In the model clean-up activity operates as additional to natural decay of the pollution stock. For example, rivers may be treated to reduce biological oxygen demand or air may be filtered to remove particles. The level of such activity will be measured by expenditure on it, V. We shall refer to V as 'clean-up expenditure'. This is a term which is widely used in the literature. Sometimes the term is used in a broad sense to include expenditure on coping with, or ameliorating the effects of, an existing level of pollution. Thus, for example, in some contexts the term would be used to cover expenditure by individuals on personal air filters, 'gas masks', for wear while walking the streets of a city with an air pollution problem. As we use the term here, it would in that context refer to expenditure on an activity intended to reduce the level of air pollution in the city.

The consequences of clean-up expenditure on the pollutant stock are described by the equation:

$$F = F(V) \tag{16.6}$$

in which $F_V > 0$. The term F, therefore, describes the reduction in the pollution stock brought about by some level of clean-up expenditure V. Incorporating this in the differential equation for the pollutant stock gives

$$\dot{A} = M(R) - \alpha A - F(V) \tag{16.7}$$

which says that the pollution stock is increased by emissions arising from resource use and is decreased by natural decay and by clean-up expenditure.[2]

16.1.5 The optimisation problem

The dynamic optimisation problem can now be stated as:

Select values for the control variables C_t, R_t and V_t for $t = 0, \ldots, \infty$ so as to maximise

$$W = \int_{t=0}^{t=\infty} U(C_t, E(R_t, A_t))e^{-\rho t}dt$$

subject to the constraints

$$\dot{S} = -R_t$$

$$\dot{A}_t = M(R_t) - \alpha A_t - F(V_t)$$

$$\dot{K} = Q(K_t, R_t, E(R_t, A_t)) - C_t - \Gamma(R_t) - V_t$$

As shown in Table 16.1, there are three state variables in this problem: S_t, the resource stock at time t; A_t, the level of ambient pollution stock at time t; and K_t, the capital stock at time t. Associated with each state variable is a shadow price, P (for the resource stock), ω (for the capital stock) and λ (for the ambient pollution stock). Note a change in notation here; in this chapter we use upper case P for the shadow price (and net price) of the resource, whereas in Chapter 15 we use lower case p to denote a net price. Be careful to note that, because we are maximising a utility-based social welfare function, the discount rate being used here is a utility discount rate (not a consumption discount rate) and the shadow prices

Table 16.1 Key variables and prices in the model

Variables ($t = 0, \ldots, \infty$)	
Instrument (control) variables:	
C_t R_t V_t	
State variables:	Co-state variables (shadow prices) ($t = 0, \ldots, \infty$)
S_t K_t A_t	P_t ω_t λ_t

[2] Note that in our formulation, clean-up consequences do not depend on A. This restriction may not be appropriate in all circumstances; if it were not, the model being used here would need to be generalised by specifying $F = F(V, A)$.

are denominated in units of utility (not in units of consumption). This should be taken into account when comparing the shadow price of the ambient pollution stock in this chapter (λ) with the shadow price μ used in Chapter 5 (which was measured in consumption units).

In the production function specified by equation 16.4 we assume that $Q_E < 0$ (and also, as before, $E_R > 0$ and $E_A > 0$). The rate of extraction of environmental resources thus has a direct and an indirect effect upon production. The direct effect is that using more resources increases Q. The indirect effect is that using more resources increases environmental pressures, and so reduces production. The overall effect of R on Q is, therefore, ambiguous and cannot be determined *a priori*.

16.1.6 The optimal solution to the model

The current-valued Hamiltonian is

$$\begin{aligned} H = {} & U(C_t, E(R_t, A_t)) + P_t(-R_t) \\ & + \omega_t(Q[K_t, R_t, E(R_t, A_t)] - C_t - \Gamma(R_t) - V_t) \\ & + \lambda_t(M(R_t) - \alpha A_t - F(V_t)) \end{aligned}$$

Ignoring time subscripts, the necessary conditions for a social welfare maximum are:[3]

$$\frac{\partial H}{\partial C} = U_C - \omega = 0$$

$$\frac{\partial H}{\partial R} = U_E E_R - P + \omega Q_R + \omega Q_E E_R - \omega\Gamma_R + \lambda M_R$$
$$= 0$$

$$\frac{\partial H}{\partial V} = -\omega - \lambda F_V = 0$$

$$\dot{P} = -\frac{\partial H}{\partial S} + \rho P \Leftrightarrow \dot{P} = \rho P$$

$$\dot{\omega} = -\frac{\partial H}{\partial K} + \rho\omega \Leftrightarrow \omega = \rho\omega - Q_K\omega$$

$$\dot{\lambda} = -\frac{\partial H}{\partial A} + \rho\lambda$$

$$\Leftrightarrow \dot{\lambda} = \rho\lambda + \alpha\lambda - U_E E_A - \omega Q_E E_A$$

These can be rewritten as:

$$U_C = \omega \tag{16.8a}$$

$$P = U_E E_R + \omega Q_R + \omega Q_E E_R - \omega\Gamma_R + \lambda M_R \tag{16.8b}$$

$$\omega = -\lambda F_V \tag{16.8c}$$

$$\dot{P} = \rho \mathrm{P} \tag{16.8d}$$

$$\dot{\omega} = \rho\omega - Q_K\omega \tag{16.8e}$$

$$\dot{\lambda} = \rho\lambda + \alpha\lambda - U_E E_A - \omega Q_E E_A \tag{16.8f}$$

16.1.7 Interpreting the solution

Three of these first-order conditions for an optimal solution – equations 16.8a, 16.8d and 16.8e – have interpretations essentially the same as those we offered in Chapter 14. No further discussion of them is warranted here, except to note that equation 16.8d is a Hotelling dynamic efficiency condition for the resource net price, which can be written as:

$$\frac{\dot{P}}{P} = \rho$$

Provided that the utility discount rate is positive, this implies that the resource net price must always grow at a positive rate. Note that the ambient pollution level does not affect the growth rate of the resource net price.

Three conditions appear that we have not seen before, equations 16.8b, 16.8c and 16.8f. The last of these is a dynamic efficiency condition which describes how the shadow price of pollution, λ, must move along an efficient path. As this condition is not central to our analysis, and because obtaining an intuitive understanding of it is difficult, we shall consider it no further. However, some important interpretations can be drawn from equations 16.8b and 16.8c. We now turn to these.

16.1.7.1 The static efficiency condition for the resource net price

Equation 16.8b gives the shadow net price of the environmental resource. It shows that the net price of the environmental resource equals the value of the

[3] We will leave you to verify that these first-order conditions are correct, using the method of the maximum principle explained in Appendix 14.1.

marginal net product of the environmental resource (that is, ωQ_R, the value of the marginal product less $\omega\Gamma_R$, the value of the extraction costs) minus three kinds of damage cost:

- $U_E E_R$, the loss of utility arising from the impact of a marginal unit of resource use on environmental pressures;
- $\omega Q_E E_R$, the loss of production arising from the impact of a marginal unit of resource use on environmental pressures;
- λM_R the value of the damage arising indirectly from resource extraction and use. This corresponds to what we have called previously stock-damage pollution damage. This 'indirect' damage cost arises because a marginal increase in resource extraction and use results in pollution emissions and then an increase in the ambient pollution level, A. To convert this into value terms, we need to multiply this by a price per unit of ambient pollution.

Note that we have stated that these three forms of damage cost must be *subtracted* from the marginal net product of the environmental resource, even though they are each preceded by an addition symbol in equation 16.8b. This can be verified by noting that U_E and Q_E are each negative, as is the shadow price λ_t, given that ambient pollution is a 'bad' rather than a 'good' and so will have a negative price.

In a competitive market economy, none of these pollution damage costs will be internalised – they are not paid by whoever it is that generates them. This has implications for efficient and optimal pollution policy. A pollution control agency could set a tax rate per unit of resource extracted equal to the value of marginal pollution damages, $U_E E_R + \omega Q_E E_R + \lambda M_R$.

The nature of the required tax is shown more clearly in Figures 16.2 and 16.3. To interpret these diagrams, it will be convenient to rearrange equation 16.8b to:

$$\omega Q_R = P + \omega\Gamma_R - U_E E_R - \omega Q_E E_R - \lambda M_R$$

We can read this as saying that:

> Gross price = net price + extraction cost + value of flow damage operating on utility + value of flow damage operating on production + value of stock damage

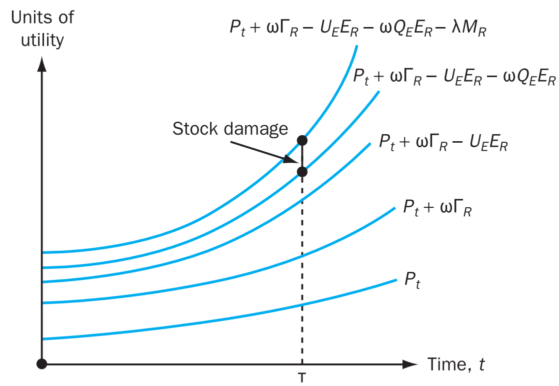


Figure 16.2 Optimal time paths for the variables of the pollution model

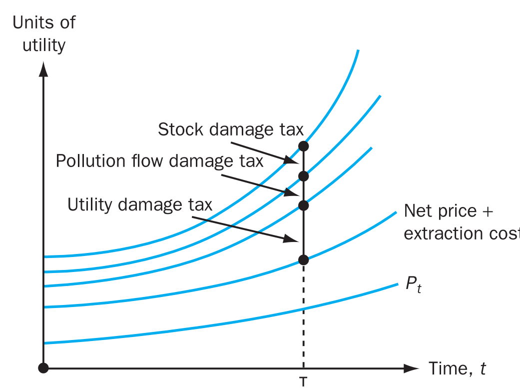
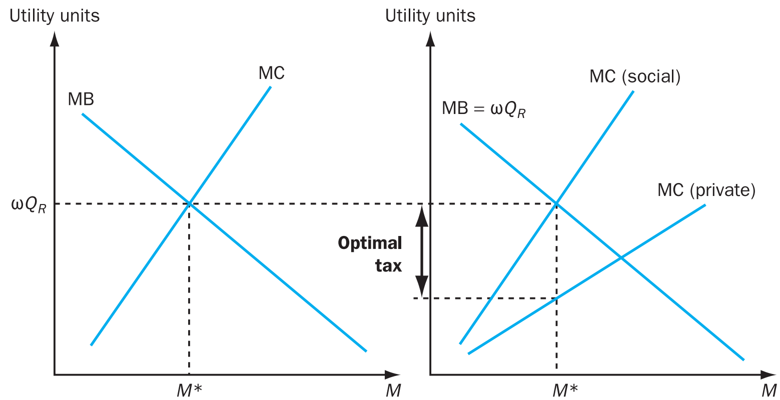


Figure 16.3 Optimal 'three-part' pollution taxes

Figure 16.2 can be interpreted in the following way. In a perfectly functioning market economy with no market failure, in which all costs and benefits are fully and correctly incorporated in market prices, the gross (or market) price of the resource would follow a path through time indicated by the uppermost curve in the diagram. We can distinguish several different cost components of this *socially efficient* gross price:

1. the net price of the resource (the rent that must be paid to the resource owner to persuade him or her to extract the resource);
2. the marginal cost of extracting the resource;
3. the marginal pollution damage cost. This consists of three different types of damage:
   - pollution flow damage operating through the utility function;



Notes: MC = MC (social) = $P + \omega\Gamma_R - U_E E_R - \omega Q_E E_R - \lambda M_R$
MC (private) = $P + \omega\Gamma_R$
Optimal tax = $-U_E E_R - \omega Q_E E_R - \lambda M_R$
MB = ωQ_R

Figure 16.4 Optimal taxes and the wedge between private and social costs

- pollution flow damage operating through the production function;
- pollution stock damage (which in our model can work through both production and utility functions).

However, in a competitive market economy where damage costs are not internalised and so do not enter firms' cost calculations, the market price will not include the pollution damage components, and so would not be equal to the gross price just described. The market price would only include two components: the net price (or resource royalty) and the marginal extraction cost. It would then be given by the curve drawn second from the bottom in Figure 16.2.

But now suppose that government were to introduce a socially optimal tax in order to bring market prices into line with the socially optimal gross price. It is now easy to see what such a tax would consist of. The tax should be set at a rate equal in value (per unit of resource) to the sum of the three forms of damage cost, thereby internalising the damages arising from resource use. We could regard this tax as a single pollution tax, or we might think of it as a three-part tax (one on utility flow damages, one on production flow damages and one on stock damages). Such an interpretation is shown in Figure 16.3. The three-part tax has the advantage that it shows clearly what the government has to calculate in order to arrive at a socially optimal tax rate.

Figure 16.4 shows this interpretation of the optimal tax rate in terms of a 'wedge' between the private and the social marginal costs. As you can see from the notes that accompany the diagram, the private marginal cost is given by $P + \omega\Gamma_R$. The optimal tax is set equal to the marginal value of the three damage costs. When imposed on firms, the wedge between private and social marginal costs is closed. Be careful to note, however, that Figure 16.4 can only be true at one point in time. We know that all the components of costs change over time, and so the functions shown in the diagram will be shifting as well.

16.1.7.2 Efficiency in clean-up expenditure

The necessary conditions for a solution of our pollution problem include one equation, equation 16.8c, that concerns clean-up expenditure, $\omega = -\lambda F_V$. To understand this condition, let us recall the meanings of its terms. First, the variable ω is the shadow price of capital; it is the amount of utility lost when one unit of output is diverted from consumption (or investment in capital) to be used for clean-up expenditure. Be careful to note that these values are being measured at the optimal solution. That is, it is the amount of utility lost when output is diverted to pollution clean-up when consumption and clean-up are already at their socially optimal levels.[4] You can imagine that finding out what these values are going to be is a very difficult task indeed; this is a matter we shall return to shortly.

Second, λ is the optimal value of one unit of ambient pollution; remember that this is a negative quantity, as pollution is harmful. Third, F_V is the amount of pollution stock clean-up from an additional unit of clean-up expenditure.

Putting these pieces together, we can deduce the meaning of equation 16.8c. The right-hand side, $-\lambda F_V$, is the utility value gained from pollution cleanup when one unit of output is used for clean-up expenditure. This must be set equal to the value of utility lost by reducing consumption (or investment) by one unit. Put in another way, the optimal amount of pollution clean-up expenditure will be the level at which the marginal costs and the marginal benefits of clean-up are equal.

16.1.7.3 An application of a stock pollution model to the problem of global climate change

During the past 30 years or so, William Nordhaus – with various collaborators – has been developing a suite of integrated economic–scientific models to examine policy responses to global climate change. While being different from the model we have developed in Section 16.1 in the way it is structured, in its degree of richness and complexity, and in many of its specifics, Nordhaus's DICE (Dynamic Integrated model of Climate and the Economy) model shares much of the essence of the model developed above. It is an optimal growth model, augmented by the presence of a rather sophisticated science component that models emissions paths into greenhouse gas atmospheric concentrations, and then on into implied time paths for global mean temperature changes. Using the latest implementation of this model, DICE-2007, Nordhaus is able to estimate a global efficient emissions reduction schedule over time, and to suggest particular rates of carbon tax that would bring about an efficient outcome.

A summary description of the DICE-2007 model, and some results obtained from it, is provided in Box 16.1. The DICE model, along with some other models of global climate change and the body of work that has emanated from the IPCC, were looked at earlier in Chapter 9.

Box 16.1 Nordhaus: The DICE-2007 model of global warming

During the past 30 years, William Nordhaus – with various collaborators – has been developing a suite of integrated economic–scientific models of global warming. The most recent version, the DICE-2007 model, is described in Nordhaus (2008), and is also available electronically at Nordhaus's personal website http://www.econ.yale.edu/~nordhaus/homepage/, together with versions of the model programmed in GAMS and in Excel. The latter is relatively easy to use.

The structure of DICE

DICE-2007 employs an optimal economic growth modelling framework, integrated with a science component which comprises a carbon cycle and a system of climate equations. The carbon cycle is based on a three-reservoir model calibrated to existing carbon-cycle models and historical data. The three reservoirs are the atmosphere, a quickly mixing reservoir in the upper oceans and the biosphere, and the deep oceans. Carbon flows in both directions between adjacent reservoirs. The mixing between the deep oceans and other reservoirs is extremely slow. The climate equations include an equation for radiative forcing and two equations for the climate system. The radiative-forcing equation calculates the impact of the accumulation of greenhouse gases (GHGs) on the radiation balance of the globe. The climate equations calculate the mean surface temperature of the globe and the average temperature of the deep oceans for each time-step. These equations draw upon and are calibrated to large-scale general circulation models of the atmosphere and ocean systems.

[4] The reason why we can use the value lost by diverting expenditure from either consumption or investment follows from this point: at the social optimum, the value of an incremental unit of consumption will be identical to the value of an incremental unit of investment. They will not be equal away from such an optimum.

Box 16.1 ***continued***

In DICE-2007 model, the only GHG that is subject to controls is industrial CO_2. This approach is taken as CO_2 is the major contributor to global warming, and because other GHGs are likely to be controlled in different ways. Other GHGs (CO_2 emissions from land-use changes, other well-mixed GHGs, and aerosols) are included as exogenous trends in radiative forcing.

CO_2 emissions depend on total output, a time-varying emissions–output ratio, and an emissions-control rate. The emissions–output ratio is estimated for individual regions, and then aggregated to the global ratio. The emissions-control rate is determined by the climate-change policy being examined. The emissions abatement cost function is parameterised by a log-linear function calibrated to recent studies of the cost of emissions reductions.

Projected damages from the economic impacts of climate change rely on estimates from earlier syntheses of the damages, with updates in light of more recent information. The basic assumption is that the damages from gradual and small climate changes are modest, but the damages rise non-linearly with the extent of climate change.

The economic growth model component of DICE is broadly similar in structure to the optimal growth model we developed in Section 16.1. However, there are some important differences. These are listed in Table 16.2 (at the foot of this box) which notes points of similarity and difference between DICE-2007 and the model you examined earlier in Section 16.1. Table 16.2 also gives information about the component parts of the DICE model, including its objective function, the production function employed, and the choice of discount rate. One other important difference is that DICE, unlike the model used in this chapter, is in discrete rather than continuous time.

The baseline case using DICE

Using consensus estimates of values for the key driving variables (such as growth of labour force, capital stock, technical change, emissions-to-output ratios, and so on) *and assuming that no significant additional emissions reductions are imposed*, one can use DICE-2007 to generate predictions about GDP, climate change, and the impacts of climate change over some suitable future span of time. These model inputs and outputs define what is called the baseline case. (Although Nordhaus does not use this term, such a baseline simulation is also known as a 'Business As Usual' case.) Some of the principal characteristics of the baseline case from a DICE-2007 simulation are:

- Rapid continued increase in CO_2 emissions from 7.4 billion tons of carbon per year in 2005 to 19 billion tons per year in 2100.
- A rapid increase in atmospheric concentrations of CO_2 from 280 parts per million (ppm) in preindustrial times to 380 ppm in 2005 and to 685 ppm in 2100.
- Measured mean global surface temperature in 2005 increased by 0.7°C relative to 1900 levels and is projected to increase by 3.1°C in 2100 relative to 1900, and by 5.3°C in 2200 relative to 1900.
- The climate changes associated with these temperature changes are estimated to increase damages by around 2.5% of global output in 2100 and by close to 8% of global output in 2200. The damages are likely to be most heavily concentrated in low-income and tropical regions such as tropical Africa and India.

Optimisation and policy analysis using DICE

One can also run DICE-2007 in full optimising mode. This results in an idealised policy that can be called the 'optimal' economic response, one that establishes a time profile of emissions control which maximises the global economic welfare function. Such a policy entails GHG emissions being reduced (below baseline levels) in a way that is efficient across industries, countries, and time. To obtain such an efficient policy outcome the marginal costs of reducing CO_2 and other GHGs should be equalised in each sector and country; furthermore, in every year the marginal abatement cost should be equal to the marginal benefit in lower future damages from climate change.

This optimal time profile of emissions control is of interest in itself. For example, an important output of the optimal policy response run of DICE-2007 is an estimate of the 'optimal carbon price,' or 'optimal carbon tax'. This is the price on carbon emissions that balances the incremental costs of reducing carbon emissions with the incremental benefits of reducing climate

Box 16.1 ***continued***

damages. However, the optimal time profile of emissions also serves as a benchmark against which various alternative policy responses to global warming can be assessed.

Major results from DICE policy analyses

A. Efficient policy response

1. Efficient emissions reductions follow a 'policy ramp' in which policies involve modest rates of emissions reductions in the near term, followed by sharp reductions in the medium and long terms. The optimal emissions-reduction rate for CO_2 relative to the baseline is 15% in the first policy period, increasing to 25% by 2050 and 45% by 2100. This path reduces CO_2 concentrations, and the increase in global mean temperature relative to 1900 is reduced to 2.6°C for 2100 and 3.4°C for 2200. (Note that these comparisons measure the emissions-reduction rates relative to the calculated baseline emissions scenario, not as with most applications relative to a historical baseline such as 1990 emissions levels.)
2. For the efficient climate-change policy, the net-present-value global benefit of the optimal policy is $3 trillion relative to no controls. This total involves $2 trillion of abatement costs and $5 trillion of reduced climatic damages.
3. The economically optimal carbon price or carbon tax would be $27 per metric ton in 2005 in 2005 prices. For carbon dioxide (smaller by a factor of 3.67) the optimal tax is $7.40 per ton of CO_2. The optimal carbon price would rise steadily over time, at a rate between 2 and 3% per year in real terms, to reflect the rising damages from climate change. In the optimal trajectory, the carbon price would rise from an initial level of $27 per ton of carbon to $90 per ton of carbon by 2050 and $200 per ton of carbon in 2100.
4. The upper limit on the carbon price is determined by the price at which all uses of fossil fuels can be economically replaced by other technologies. This is cost of the backstop technology. DICE estimates this to have an upper limit of around $1000 per ton of carbon over the next half-century or so, falling thereafter at an unknown rate.

B. Alternative policy responses

5. For most of the climatic-limits cases examined (such as adding a constraint that limits the atmospheric concentration of CO_2 to two times its pre-industrial level, or adding a constraint that limits the global temperature increase to 2.5°C), DICE finds that the net present value of global economic welfare is close to that of the optimal case. Moreover, the near-term carbon taxes that would apply to the climatic limits, except for the most stringent cases, are close to that of the economic optimum. For example, the 2005 carbon prices associated with CO_2 doubling and the 2.5°C increase are $29 and $31 per ton of carbon, respectively, compared with $27 per ton for the pure optimum without climatic limits.
6. All existing programmes fail to meet the efficiency criteria. In particular, the Kyoto Protocol is seriously flawed in terms of efficiency outcomes, being only about 0.02 as effective as the optimal policy in reducing climatic damages and still incurring high abatement costs. It is likely to be ineffective even with US participation.
7. DICE modelling results point to the importance of near-universal participation in programmes to reduce greenhouse gases. Because of the structure of the costs of abatement, with marginal costs being very low for the initial reductions but rising sharply for higher reductions, there are substantial excess costs if the preponderance of sectors and countries are not fully included. A participation rate of 50%, as compared with 100%, will impose an abatement-cost penalty of 250%. Even with the participation of the top 15 countries and regions, consisting of three-quarters of world emissions, we estimate that the cost penalty is about 70%.
8. A low-cost and environmentally benign substitute for fossil fuels would be highly beneficial. A low-cost zero-carbon technology would have a net value of around $17 trillion in present value because it would allow the globe to avoid most of the damages from climate change. No such technology presently exists, none of the mooted options is currently feasible, but the net benefits of zero-carbon substitutes are so high as to warrant very intensive research.

RICE

A regionally disaggregated version (of a previous version of DICE) has also been produced by

Box 16.1 ***continued***

Table 16.2 A comparison between the DICE model and the dynamic pollution model of Section 16.1

Component	Model in Section 16.1	DICE-2007
Objective function	$w = \int_{t=0}^{t=\infty} U(C_t, E(R_t, A_t))e^{-\rho t}dt$	DICE has a similar objective function, but one different in two principal ways. First, environmental quality does not enter the objective function explicitly, but is treated instead as part of consumption, C. Second, the objective function is specified in terms of consumption rather than utility. It is increasing in the per capita consumption of each generation, with diminishing marginal utility of consumption. The importance of a generation's per capita consumption depends on the size of the population. The pure rate of time preference and the elasticity of the marginal utility of consumption interact to determine the discount rate on goods. These parameters are set to be consistent with observed economic outcomes as reflected by interest rates and rates of return on capital.
Control (instrument) variables	C_t, R_t and V_t for $t = 0, \ldots, \infty$	DICE does not deal with clean-up expenditure as such, treating it instead as part of a separate abatement costs function. The economy has two major decision variables in the model: the overall savings rate for physical capital (which is equivalent to treating C as a control variable); and the emissions-control rate for greenhouse gases (rather than having R as a control variable).
Resource stock constraint	$\dot{S}_t = -R_t$	DICE recognises the finiteness of fossil-fuel stocks. Carbon fuels are limited in supply. Substitution of non-carbon fuels for carbon fuels takes place over time as carbon-based fuels become more expensive, either because of resource exhaustion or because policies are taken to limit carbon emissions. As carbon fuel stocks are increasingly depleted over time, the shadow price of carbon rises along a Hotelling-type path. DICE explicitly includes a backstop technology for non-carbon energy, which allows for the complete replacement of all carbon fuels at a price that is relatively high but declines over time.
Pollution stock–flow relationship	$\dot{A}_t = M(R_t) - \alpha A_t - F(V_t)$	DICE does not deal with clean-up expenditure as such, nor does it assume a constant decay parameter.
Production function	$Q = Q(K_t, R_t, E(R_t, A_t))$	DICE does not include E in the production function. Output is produced by a Cobb–Douglas production function (PF) in capital, labour (proportional to population), and energy. Energy takes the form of either carbon-based fuels or non-carbon-based technologies (such as solar, geothermal or nuclear). Technological change takes two forms: economy-wide technological change and carbon-saving technological change. Carbon-saving technological change is modelled as reducing the ratio of CO_2 emissions to output. Population growth is exogenous. The global PF is aggregated from region-specific PFs, which allow for varying rates of technological change. Aggregation uses purchasing-power parity (PPP) exchange rates.
Capital accumulation	$\dot{K}_t = Q_t - C_t - \Gamma(R_t) - V_t$	The final two terms in the capital accumulation equation do not appear in the DICE equivalent (but are dealt with elsewhere in the model).

Box 16.1 *continued*

Nordhaus and colleagues. RICE (Regional Integrated model of Climate and the Economy) is described in Nordhaus and Boyer (1999). This model was discussed in the third edition of this textbook, and the relevant box remains available on the Companion website for interested readers. A new version of RICE, consistent with DICE-2007, became available in 2010. Details of this updated version can be found via a link from Nordhaus's home page at http://www.econ.yale.edu/~nordhaus/homepage/RICEmodels.htm.

16.2 A complication: variable decay of the pollution stock

Throughout this chapter, we have assumed that the proportionate rate of natural decay of the pollution stock, α, is constant. Although a larger amount of decay will take place the greater is the size of the pollution stock, the proportion that naturally decays is unaffected by the pollution stock size (or by anything else). This assumption is very commonly employed in environmental economics analysis.

However, this assumption is usually made for convenience and analytical simplicity. Whether it is reasonable or not depends on the problem under study. Often it will not be reasonable, because the rate of decay changes over time, or depends on the size of the pollution stock. Of particular importance are the existence of threshold effects, irreversibilities, and time lags in flows between various environmental media (as with greenhouse gases). For an example of threshold effects and irreversibilities, consider river pollution. At some threshold level of biological oxygen demand (BOD) on a river, the decay rate of a pollutant may collapse to zero. An irreversibility exists if the decay rate of the pollutant in the environmental medium remains below its previous levels even when the pollutant stock falls below the threshold level. Irreversibility implies some hysteresis in the environmental system: the history of pollutant flows matters, and reversing pollution pressures does not bring one back to the *status quo ex ante*.

Another way of thinking about this issue is in terms of carrying capacities (or assimilative capacities, as they are sometimes called) of environmental media. In the case of water pollution, for example, we can think of a water system as having some pollution-assimilative capacity. This enables the system to carry and to continuously transform some proportion of these pollutants into harmless forms through physical or biological process. Our model has in effect assumed unlimited carrying capacities: no matter how large the load on the system, more can always be transformed at a constant proportionate rate.

Whether this assumption is plausible is, in the last resort, an empirical question. However, there are good reasons to believe that it is not plausible for many types of pollution. The DICE-2007 model, for example, (see Box 16.1) employs a three-box carbon-cycle model to examine carbon flows within and between the three main carbon sinks: the atmosphere; the upper oceans/biosphere; and the deep oceans. Current scientific knowledge has established that it is inappropriate to specify a constant decay rate of atmospheric CO_2 (or indeed for any of the three sinks), and so the DICE model does not impose constancy of decay rates in any part of its carbon-cycle model.

Where the assumption of a constant decay rate is grossly invalid, it will be necessary to respecify the pollutant stock–flow relationship in an appropriate way. Suggestions for further reading at the end of this chapter point you to some literature that explores models with variable pollution-decay rates.

16.3 Steady-state outcomes

Article 2 of the United Nations Framework Convention on Climate Change (UNFCCC) states that 'The ultimate objective of this convention . . . is to achieve . . . stabilization of GHG concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system'.[5] There is no consensus as to what this level is, nor perhaps could there be, given the judgement that inevitably must surround the word 'dangerous'. A widely held opinion among natural and physical scientists is that the target should be set at twice the pre-industrial concentration (i.e. at 560 ppm of CO_2 or 1190 GtC in the atmosphere). Many climate-change research teams have employed this value for one of the scenarios they have investigated. As we mentioned in Box 16.1, Nordhaus's DICE model simulations now suggest that imposing such a doubling as a constraint on our emissions behaviour is close to being economically optimal.

More recently, a growing number of climate scientists are warning that if temperatures were to rise more than 2°C above 1900 mean levels, a 'tipping point' would be passed beyond which positive feedback effects would generate runaway warming, possibly leading to catastrophic outcomes.[6] Many climate change modellers are now also looking at what kinds of emissions control programmes would satisfy the constraint that mean global temperature increases do not exceed that particular level.[7]

Both the 'doubling of carbon dioxide concentration' and '2°C' constraints may be seen as examples of the notion that policy should be constrained by some form of precautionary principle. In thinking about the greenhouse effect, using the precautionary principle suggests that policy maker should try to identify what kinds of states are acceptable in terms of avoiding risks of catastrophic climate change, and rule out any policy that has a reasonable likelihood of pushing an economy into any unacceptable state.[8]

At first blush, policies that argue for constant levels of something – atmospheric CO_2 concentration, or global mean temperature, in the examples just mentioned – might be thought of as advocating steady-state outcomes.[9] However, that is not the case. As you have seen in some of the previous chapters (for example, Chapter 5 in discussing pollution targets) where we did examine steady-state outcomes, those are defined as equilibria in which the levels of *all* state variables in a system are unchanging through time.[10]

Is the notion of a steady state useful, or even meaningful, in the context of the modelling framework we have been examining in Section 16,1, in which a long-lived stock pollutant is emitted as the result of combustion of a fossil fuel that is a non-renewable resource for which there is a finite total stock? Intuition suggests that it may not be. Indeed, looking back at the various four-quadrant diagrams in the previous chapter, it is evident that nothing resembling a steady state is attained until the non-renewable resource is fully depleted, or until that resource ceases to be used for some reason other than simple exhaustion of the resource. Furthermore, our analysis in Chapter 14 demonstrated that if there is no substitute for such a non-renewable resource *and* that resource is essential in production, then a steady state will not be reached in any finite span of time. Thus, for the pollution model we have just been studying in section 16.1, the notion of a steady state is logically inconsistent and so not meaningful. No constant positive quantity of a non-renewable resource can be extracted indefinitely, given the limited stock size. The only constant amount that could be used indefinitely is zero. That case is of no interest in our model as it stands. For if R were zero, production would be using produced capital as the sole input. That is at odds with the laws of thermodynamics: unless the capital itself was consumed in

[5] The full wording of the Convention can be found at http://unfccc.int.

[6] The figure of 2°C (and the reference date of 1900) are by no means universally agreed on. Suffice it to say that these figures are being widely discussed in the literature now, and are not far from the alternative figures that others who are concerned about such a tipping point have proposed.

[7] Nordhaus's DICE model simulations also suggest that imposing this constraint on our emissions behaviour is close to being economically optimal.

[8] We also note that another important justification for the use of a precautionary principle is the existence of uncertainty about parameter values and the true structure of the relevant model of the process involved. If one knew with certainty both the values of all relevant parameters and the structure of the model then one might be justified in terms of dealing only with model-deduced optimal outcomes. But where there is uncertainty in regard to either of these matters, there is a strong case for embedding policy within a precautionary principle framework.

[9] Of course, this is not actually what those policies implied. Rather, these were being treated as upper limits that should not be breached. In other words, they were constraints rather than objectives in themselves.

[10] A more general notion of steady-state equilibrium is one in which all state variables are growing over time at the same fixed rate, g. The concept we have been using in this book is the special case in which g is zero.

the process of production, it implies that physical output could be produced without using any physical inputs, which is clearly impossible.

It seems rather implausible, though, to imagine that fossil fuels (or indeed any other non-renewable resource) have zero substitution possibilities. So if we wish to think about policy over indefinitely long spans of time, it is necessary to consider substitutes for non-renewables. Two alternatives suggest themselves.

First, there is the category of backstop technologies. As we have examined the backstop case at some length in Chapter 15 we shall not explore it any further here. But note, from Box 16.1, that Nordhaus's DICE model has now incorporated a backstop technology in its framework, and estimates that a carbon price of about US$1000 (in 2005 price terms) would be sufficient to make *all* uses of fossil fuels economically unviable. Of course, many uses would be uneconomic at carbon prices well below that figure, and some even at current prices of carbon permits on the EU ETS.

A second possibility is that economic activity might increasingly switch to using renewable resources in place of non-renewables.[11] It is evident that production cannot rely forever only on the use of non-renewable resources. At some point in time it will be necessary to make use of renewable resources as productive inputs. This points us to a way in which the model investigated above should be extended if it is to be useful for very long-term analysis. And as we shall see (in Chapter 17), it also leads to the notion of a steady state in which the renewable resource is used at a constant rate through time being a meaningful and relevant concept.

That suggests that one might generalise the model specified in Section 16.1 to a version in which the production function is of the form $Q = Q(R_1, R_2, K, E)$ where R_1 and R_2 denote non-renewable and renewable natural resources respectively. Here R_2 might as a heterogeneous collection of renewables, including renewable flow energy resources (such as wind, wave, solar and geothermal energy), elements of which become economically more attractive than fossil fuels when fossil-fuel prices reach particular levels. One then finds a sequence of substitutions taking place (as if there is a series of backstops rather than just one), until some fossil-fuel price is reached at which backstops have completely taken over. We leave it to you as an exercise to investigate how the model we have been examining could be generalised in this way, and what its steady state would consist of. A possible answer is provided in the file *Enlarged model.doc* in the *Additional Materials* for Chapter 16.

16.4 A model of waste accumulation and disposal

In this section, we investigate efficient emissions targets for stock pollutants where it is not necessary to take account of resource stock-flow constraints in the way we did in Section 16.2. Ignoring such constraints may be appropriate where pollution derives from the extraction and use of some resource on a scale sufficiently small that a resource stock constraint is not binding. We might call these 'local' models of stock pollution. These are typically more disaggregated than the models previously examined in this chapter. Examples include models of pollution associated with the use of nitrates in agricultural chemicals, with discharges of toxic substances and radioactive substances, and with various forms of groundwater and marine water contamination.

The models we are looking at here are best thought of as examples of partial equilibrium cost–benefit analysis, albeit in a dynamic modelling framework. Because variables are now being measured in consumption units rather than in utility units the

[11] In fact, a backstop energy technology can be thought of as a form of renewable energy. The term backstop energy technology refers to a limitlessly available energy source that can always be used in place of carbon-based energy, and that has little meaning outside the context of renewable 'flow' energy sources, such as solar or wind energy, and possibly renewable 'stock' energy sources (such as alcohol derived from biomass). The case of nuclear energy, which is often advocated as a backstop energy technology, is more problematic. Nuclear fission power (the type actually generated today) does not really constitute a backstop as it depends on stocks of uranium, itself a non-renewable resource. Fusion power, should it ever prove to be a feasible large-scale energy technology, would be an available backstop (although not necessarily the lowest-cost one).

appropriate rate of discount is now r rather than ρ. We shall pay particular attention to the economically efficient steady-state outcome.

To fix ideas, have in mind the nitrate pollution example.[12] We suppose that there are benefits in terms of increased agricultural yields from using nitrate-based fertilisers. However, the application of nitrate fertilisers cannot be perfectly controlled, in the sense that only the precise amount taken up by the growing biomass is applied. The surplus-to-growth applications of fertiliser will eventually contaminate groundwater resources, or will via water runoff processes be transported into other local water systems, which will themselves be contaminated. Denote the quantities of fertiliser 'emissions' as M. Let $B(M)$ denote the benefits of fertiliser use, where the interpretation of $B(M)$ is equivalent to that used in our models of pollution targets and instruments in Chapters 5 and 6. Let the economic value of damages, expressed in terms of lost consumption-equivalent possibilities, be denoted by the symbol D. However, as we are here dealing with stock pollutants, the damage caused by excess fertiliser applications depends on the concentration rate of the pollutant in relevant environmental receptors. Denoting that concentration rate by A, we thus write damages as $D(A)$. Finally, the stock–flow relationship between M and A will be modelled by a relationship of the form we have used several times previously, $\mathrm{d}A_t/\mathrm{d}t = M_t - \alpha A_t$, where α is the decay rate of the pollutant.[13]

The quantity that we seek to maximise here is the net present value of benefits less damages over some suitably defined interval of time (where that interval may be infinitely long). Box 16.2 lays out the problem we shall be considering, states the Hamiltonian for the problem, and lays out the first-order conditions for its solution.

Box 16.2 The local stock pollution model

The problem

The objective is to choose a sequence of pollutant emission flows, M_t, $t = 0$ to $t = \infty$, to maximise

$$\int_{t=0}^{t=\infty} (B(M_t) - D(A_t))\mathrm{e}^{-rt}\mathrm{d}t$$

subject to

$$\frac{\mathrm{d}A_t}{\mathrm{d}t} = M_t - \alpha A_t \tag{16.9}$$

$A_0 = A(0)$, a non-negative constant

$M_t \geq 0$

Optimisation conditions

The current-valued Hamiltonian for this problem is

$$H_t = B(M_t) - D(A_t) + \mu_t(M_t - \alpha A_t) \tag{16.10}$$

The necessary first-order conditions for a maximum (assuming an interior solution) include:

$$\frac{\partial H_t}{\partial M_t} = 0 \Rightarrow \frac{\mathrm{d}B_t}{\mathrm{d}M_t} + \mu_t = 0 \tag{16.11}$$

$$\frac{\mathrm{d}\mu}{\mathrm{d}t} = r\mu_t - \frac{\partial H_t}{\partial A_t} + r\mu_t + \frac{\mathrm{d}D}{\mathrm{d}A_t} + \alpha\mu_t \tag{16.12}$$

16.4.1 The steady state

In a steady state, all variables are constant over time and so $\mathrm{d}\mu/\mathrm{d}t$ in equation 16.12 will be zero. Also, time subscripts are now redundant and so no longer necessary. The two first-order conditions become

$$\frac{\mathrm{d}B}{\mathrm{d}M} = -\mu \tag{16.13}$$

$$\frac{\mathrm{d}D}{\mathrm{d}A} = -(r + \alpha)\mu \tag{16.14}$$

Also, in a steady state the pollution-stock differential equation

$$\frac{\mathrm{d}A_t}{\mathrm{d}t} = M_t - \alpha A_t$$

[12] One might object that this is not a good example, given the current and future projected scale of global use of nitrate-based fertilisers. The magnitudes are probably not consistent with the assumption that the resource is being used on a scale sufficiently small that a resource stock constraint is not binding.

[13] Note that we have earlier used the notation $\dot{A}$ rather than the alternative notation $\mathrm{d}A/\mathrm{d}t$. These two forms of notation are alternative ways of referring to the derivative with respect to time of the variable A.

collapses to

$$M = \alpha A \tag{16.15}$$

Equation 16.14 can also be written as

$$-\mu = \frac{\frac{\mathrm{d}D}{\mathrm{d}A}}{r + \alpha} \tag{16.16}$$

The variable μ is the shadow price of one unit of pollutant emissions. It is equal to the marginal social value of a unit of emissions at a social net benefit maximum. As pollution is a bad, not a good, the shadow price, μ, will be negative (and so –μ will be positive).

The conditions 16.13 and 16.16 say that two things have to be equal to –μ at a net benefit maximum. Therefore those two things must be equal to one another. Combining those conditions we obtain:

$$\frac{\mathrm{d}B}{\mathrm{d}M} = \frac{\frac{\mathrm{d}D}{\mathrm{d}A}}{r + \alpha} \tag{16.17}$$

Equation 16.17 is one example of a familiar marginal condition for efficiency: in this case, an efficient solution requires that the present value of net benefit of a marginal unit of pollution equals the present value of the loss in future net benefit that arises from the marginal unit of pollution. However, it is quite tricky to get this interpretation from equation 16.17, so we shall take you through it in steps.

The term on the left-hand side of equation 16.17 is the increase in *current* net benefit that arises when the rate of emissions is allowed to rise by one unit. This marginal benefit takes place in the current period only. In contrast, the right-hand side of equation 16.17 is the present value of the loss in future net benefit that arises when the output of the pollutant is allowed to rise by one unit. Note that d*D*/d*A* itself lasts for ever; it is a form of perpetual annuity (although an annuity with a negative effect on utility). To obtain the present value of an annuity, we divide its annual flow, d*D*/d*A*, by the relevant discount rate, which in this case is *r*. The reason why we *also* divide the annuity by α is because of the ongoing decay process of the pollutant. If the pollutant stock were allowed to rise, then the amount of decay in steady state will also rise by a proportion α of that increment in the stock size. This reduces the magnitude of the damage. Note that α acts in an equivalent way to the discount rate. The greater is the rate of decay, the larger is the 'effective' discount rate applied to the annuity and so the smaller is its present value.

For the purpose of looking at some special cases of equation 16.17, it will be convenient to rearrange that expression as follows:

$$\frac{\mathrm{d}D}{\mathrm{d}A} = \frac{\mathrm{d}B}{\mathrm{d}M}\alpha + \frac{\mathrm{d}B}{\mathrm{d}M}r \tag{16.18}$$

and so

$$\frac{\mathrm{d}D}{\mathrm{d}A}\frac{1}{\alpha} = \frac{\mathrm{d}B}{\mathrm{d}M} + \frac{\mathrm{d}B}{\mathrm{d}M}\frac{r}{\alpha} \tag{16.19}$$

Given that in steady state $A = (1/\alpha)M$, then from the damage function $D = D(A)$, and using the chain rule of differentiation, we can write

$$\frac{\mathrm{d}D}{\mathrm{d}M} = \frac{\mathrm{d}D}{\mathrm{d}A} \cdot \frac{\mathrm{d}A}{\mathrm{d}M} = \frac{\mathrm{d}D}{\mathrm{d}A}\frac{1}{\alpha}$$

This allows us to write equation 16.19 as

$$\frac{\mathrm{d}D}{\mathrm{d}M} = \frac{\mathrm{d}B}{\mathrm{d}M} + \frac{\mathrm{d}B}{\mathrm{d}M}\frac{r}{\alpha}$$

or

$$\frac{\mathrm{d}D}{\mathrm{d}M} = \frac{\mathrm{d}B}{\mathrm{d}M}\left(1 + \frac{r}{\alpha}\right) \tag{16.20}$$

If we knew the values of the parameters α and *r*, and the functions d*B*/d*M* and d*D*/d*M* (or d*D*/d*A*, from which d*D*/d*M* could be derived for any given value of α), equation 16.20 could be solved for the numerical steady-state solution value of *M*, *M**. Then from the relationship $A = (1/\alpha)M$ the steady-state solution for *A* is obtained, *A**.

Four special cases of equation 16.20 can be obtained, depending on whether $r = 0$ or $r > 0$, and on whether $\alpha = 0$ or $\alpha > 0$. These were laid out in Table 6.4 in Chapter 6. We briefly summarise here our earlier conclusions.

Case A: $r = 0$, $\alpha > 0$

Given that $\alpha > 0$, the pollutant is imperfectly persistent and eventually decays to a harmless form. With

$r = 0$, no discounting of costs and benefits is being undertaken. Equation 16.20 collapses to:[14]

$$\frac{dD}{dM} = \frac{dB}{dM} \tag{16.21}$$

An efficient steady-state rate of emissions for a stock pollutant requires that the contribution to benefits from a marginal unit of pollution flow be equal to the contribution to damage from a marginal unit of pollution flow. We can also write this expression as

$$\frac{dD}{dA}\frac{1}{\alpha} = \frac{dB}{dM} \tag{16.22}$$

which says that the contribution to benefits of a marginal unit of emissions flow should be set equal to the damage caused by an additional unit of ambient pollutant stock divided by α.

Case C: $r > 0$, $\alpha > 0$

Equation 16.20 remains unchanged here:

$$\frac{dD}{dM} = \frac{dB}{dM}\left(1 + \frac{r}{\alpha}\right)$$

The marginal equality we noted in Case A remains true but in an amended form (to reflect the presence of discounting at a positive rate). Discounting, therefore, increases the steady-state level of emissions. Intuitively, the reason it does so is because a larger value of r reduces the present value of the future damages that are associated with the pollutant stock. In effect, higher weighting is given to present benefits relative to future costs the larger is r. However, the shadow price of one unit of the pollutant emissions becomes larger as r increases.

Cases B ($r > 0$, $\alpha = 0$) and D ($r = 0$, $\alpha = 0$)

Given that $\alpha = 0$, cases B and D are each one in which the pollutant is perfectly persistent – the pollutant does not decay to a harmless form. No positive and finite steady-state level of emissions can be efficient. The only possible steady-state level of emissions is zero. If emissions were positive, the stock would increase without bound, and so stock-pollution damage would rise to infinity. The steady-state equilibrium solution for any value of r when $\alpha = 0$, therefore, gives zero pollution. The pollution stock level in that steady state will be whatever level A had risen to by the time the steady state was first achieved, say time T. Pollution damage continues indefinitely, but no additional damage is being caused in any period.

This is a very strong result – any activity generating perfectly persistent pollutants that lead to any positive level of damage cannot be carried on indefinitely. At some finite time in the future, a technology switch is required so that the pollutant is not emitted. If that is not possible, the activity itself must cease. Note that even though a perfectly persistent pollutant has a zero natural decay rate, policy makers may be able to find some technique by which the pollutant may be artificially reduced. This is known as clean-up expenditure. We examined this possibility in Section 16.1.4.

16.4.2 Dynamics

The previous subsection outlined the nature of the steady-state solution to the local stock pollution model. However, without some form of policy intervention, it is very unlikely that variables will actually be at their optimal steady-state levels. How could the policy maker 'control' the economy to move it from some arbitrary initial position to its optimal steady state? To answer this question, we need to carry out some analysis of the dynamics of the model solution. Our interest is with the dynamics of the state variable (A_t) and the instrument or control variable (M_t) in our problem. Specifically, we are looking for two differential equations of the form:

$$\frac{dA}{dt} = f(A, M)$$

$$\frac{dM}{dt} = g(A, M)$$

[14] We can arrive at this result another way. Recall that $NB(M) = B(M) - D(A)$. Maximisation of net benefits requires that the following first-order condition is satisfied: $dNB/dM = dB/dM - dD/dM = 0$. Differentiating (using the chain rule in the damage function) and then rearranging, we obtain $dB/dM = (1/\alpha)(dD/dA) = dD/dM$.

We already have the first of these – it is given by equation 16.9, the pollution stock–flow relationship. To obtain the second of this pair of differential equations we proceed as follows. First, take the time derivative of equation 16.11, yielding:

$$\frac{\mathrm{d}\mu}{\mathrm{d}t} = -\left(\frac{\mathrm{d}^2 B}{\mathrm{d}M^2}\right)\frac{\mathrm{d}M}{\mathrm{d}t} \tag{16.23}$$

Then substituting equation 16.23 into equation 16.12 we have:

$$(r + \alpha)\mu + \frac{\mathrm{d}D}{\mathrm{d}A} = -\left(\frac{\mathrm{d}^2 B}{\mathrm{d}M^2}\right)\frac{\mathrm{d}M}{\mathrm{d}t} \tag{16.24}$$

Finally, substituting equation 16.11 into equation 16.24 yields the second differential equation we require:[15]

$$\frac{\mathrm{d}M}{\mathrm{d}t} = \frac{(r+\alpha)\left(\dfrac{\mathrm{d}B}{\mathrm{d}M}\right) - \dfrac{\mathrm{d}D}{\mathrm{d}A}}{\dfrac{\mathrm{d}^2 B}{\mathrm{d}M^2}} \tag{16.25}$$

The differential equations 16.25 and 16.9 will provide the necessary information from which the efficient time paths of $\{M_t, A_t\}$ can be obtained. In the absence of particular functions, the solutions can only be qualitative. However, if we select particular functions and parameter values, then a quantitative solution can be obtained. In the example which follows, we choose the functions and parameter values used earlier in the text, in Box 5.7 of Chapter 5. There we had $\alpha = 0.5$, $r = 0.1$, $D = A^2$ and $B = 96M - 2M^2$, and so $\mathrm{d}B/\mathrm{d}M = 96 - 4M$, $\mathrm{d}D/\mathrm{d}A = 2A$ and $\mathrm{d}D/\mathrm{d}M = 8M$ (in steady state).

It will be convenient to obtain the steady-state solution before finding the dynamic adjustment path. Inserting the function and parameter values given in the previous paragraph into the differential equations 16.9 and 16.25 gives

$$\frac{\mathrm{d}A}{\mathrm{d}t} = M - 0.5A \tag{16.9$'$}$$

$$\frac{\mathrm{d}M}{\mathrm{d}t} = \frac{0.6(96 - 4M) - 2A}{-4} = 0.6M + 0.5A - 14.4 \tag{16.25$'$}$$

In steady state, variables are unchanging through time, so $\mathrm{d}A/\mathrm{d}t \equiv \dot{A} = 0$ and $\mathrm{d}M/\mathrm{d}t \equiv \dot{M} = 0$. Imposing these values and solving the two resulting equations yields $M^* = 9$ and $A^* = 18$ (as we found previously). This steady-state solution is shown in the 'phase plane' diagram, Figure 16.5. The intersection of the two lines labelled $\dot{A} = 0$ and $\dot{M} = 0$ (which are here $A = 2M$ and $A = (-0.6/0.5)M + 28.8$ from 16.9′ and 16.25′) gives $M^* = 9$ and $A^* = 18$.

Next, we establish in which direction A and M will move over time from any pair of initial values $\{A_0, M_0\}$. The two lines $\dot{A} = 0$ and $\dot{M} = 0$ (known as isoclines) divide the space into four quadrants. Above the line $\dot{A} = 0$, $A > 2M$, decay exceeds emissions flows, and so A is falling. Conversely, below the line $\dot{A} = 0$, $A < 2M$, decay is less than emissions flows, and so A is rising. These movements are shown by the downward-facing directional arrows in the two quadrants labelled **a** and **b**, and by upward-facing directional arrows in the two quadrants labelled **c** and **d**.

Above the line $\dot{M} = 0$, $0.6M > 14.4 - 0.5A$, and so from equation 16.25′ we see that M is rising. Below the line $\dot{M} = 0$, $0.6M < 14.4 - 0.5A$, and so M is falling. These movements are shown by the leftward-facing directional arrows in the two quadrants labelled **a** and **d**, and by rightward-facing directional arrows in the two quadrants labelled **b** and **c**.

Taking these results together we obtain the pairs of direction indicators for movements in A and M for each of the four quadrants when the system is not in steady state. The curved and arrowed solid lines illustrate four paths that the variables would take from particular initial values. Thus, for example, if the initial values in quadrant **d** with $M = 15$ and $A = 2$, the differential equations which determine A and M would at first cause M to fall and A to rise over time. As this trajectory crosses the $\dot{M} = 0$ isocline into quadrant *c*, A will continue to rise but now M will also rise too. Left alone, the system would not reach the steady-state optimal solution, diverging ever further from it as time passes.

Inspection of the other three trajectories shows that these also fail to attain the steady-state optimum, and eventually diverge from it. Indeed, there are

[15] Notice that D and B are functions of M or A.

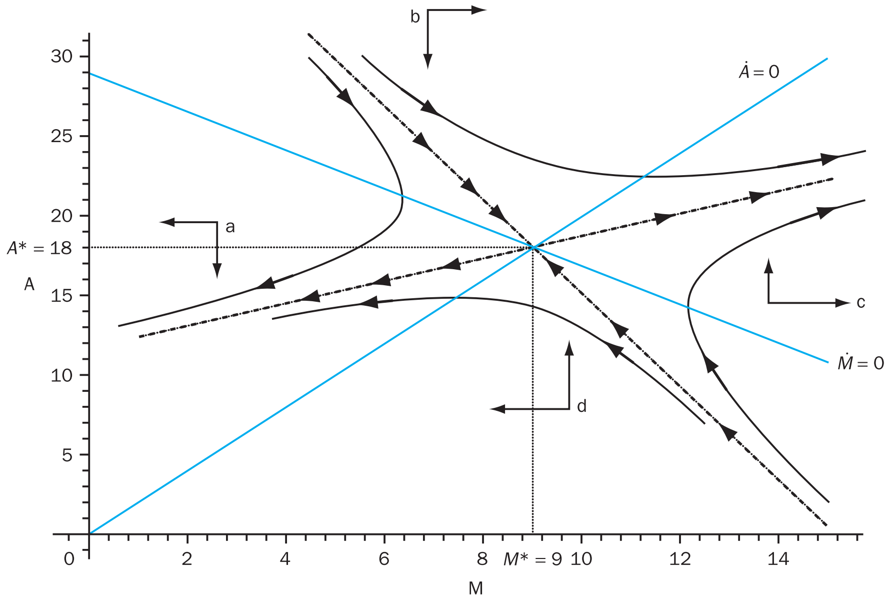


Figure 16.5 Steady-state solution and dynamics of the waste accumulation and disposal model

only two paths which do lead to that optimum. These are shown by the dashed line running from upper left towards lower right, the arrows of which point toward the steady-state solution, often referred to as the bliss point. The dashed line is the stable arm of the problem. Notice that there is also an unstable arm on which system dynamics drive away from the equilibrium. For any dynamic process with a saddle-point equilibrium such as this, the only way of reaching the optimum is for the policy maker to control M so as to reach the stable arm, and then to adjust M accordingly along the stable arm until the bliss point is reached.[16]

From all of this we have the following conclusion. If the initial level of pollution stock lies to the left of the stable arm, emissions should be increased until they reach the level indicated by the stable arm (for that level of pollution stock). The pollution stock will then rise (fall) if A_0 were less (more) than A^*, and the policy maker would need to decrease (increase) emissions to stay on the stable path until the bliss point was reached.

There are several instruments by means of which the environmental protection agency could control emissions in this way. For example, it could issue quantity regulations (by issue of licences); it could use a marketable permit system; or it could use an emissions tax or abatement subsidy. Note that the regulator will need to keep in mind both the steady-state solution which it wishes to be ultimately achieved, and the transition path to it. For the latter purpose, regulation will typically change in severity over time if an optimal approach to the equilibrium is to be achieved.

[16] A useful intuitive explanation of a saddle point equilibrium is found on Wikipedia (by searching under 'Saddle point'). The page states that: 'A saddle point is a point in the domain of a function which is a stationary point but not a local extremum. The name derives from the fact that in two dimensions the surface resembles a saddle that curves up in one direction, and curves down in a different direction (like a horse saddle or a mountain pass). In terms of contour lines, a saddle point can be recognized, in general, by a contour that appears to intersect itself. For example, two hills separated by a high pass will show up a saddle point, at the top of the pass, like a figure-eight contour line.'

A good way of visualising a saddle point is by constructing a 3d plot of the function $z - x^2 - y^2$ (in which, say, both x and y range over the interval –10 to +10. You will observe a saddle point at the point where $x = y = 0$.

By the term 'bliss point' we mean here a local extremum.

In the steady state, the terminal condition (transversality condition) will be satisfied. $A_{t=T} = A^*$ and $M_{t=T} = M^*$. Here $M_t = M_T = \alpha A_T$ so that $\mathrm{d}A/\mathrm{d}t = 0$, and the pollution stock remains at the steady-state level. The terminal conditions for pollution emissions are $M_T = \alpha A_T$ from equation 16.9 and, from equation 16.17,

$$(r + \alpha)\frac{\mathrm{d}B(M_T)}{\mathrm{d}M_T} - \frac{\mathrm{d}D(A_T)}{\mathrm{d}A_T} = 0$$

If the reader would like to see in more detail how these properties can be discovered using a computer software package, we suggest you examine the Maple file *Stock pollution 1.mw*. This file is set up to generate the picture reproduced here as Figure 16.5. For a much more extensive account of the techniques of dynamic analysis using phase-plane diagrams, see the file *Phase.doc*. Both of these are available in the *Additional Materials* for Chapter 16.

Summary

In this chapter we have developed and studied two further models of optimal emissions. Each of these is a variant of the optimal growth modelling framework used before at several places in the text. The two models developed here are appropriate for the specific circumstance in which emissions result in the accumulation of persistent (long-lasting) pollutants. They have the attractive property that choices about resource use and choices about pollution control are brought together in one integrated modelling exercise.

The first of the two models discussed in this chapter has been called an aggregate stock pollution model. This model is particularly useful for dealing with national, international or global pollution problems arising from the extraction or use of fossil fuels, and where the pollutant is long-lasting and accumulates over time. In effect, what we have done within this modelling framework is to show how optimal pollution targets can, at least in principle, be obtained from generalised versions of the resource depletion models we investigated in Chapters 14 and 15. There are, of course, severe practical problems in implementing such models, particularly given the conditions of limited information and uncertainty in which researchers and policy makers must operate.

The second model investigated here – what we termed ‘the model of waste accumulation and disposal’ – showed how one might think about the setting of emissions targets for various stock pollution problems that are more local, or less pervasive, than those we addressed with the previous model. Examples of these types of problems would include the accumulation of lead in water systems, contamination of water supplies by agricultural pesticide use, and contamination of water systems by effluent discharges.

In all modelling exercises of the kind developed in this chapter, the time dimension is of the essence. Optimal (or merely efficient) pollution targets are not necessarily constant over time, even where flows are unchanging over time. Rather, targets should be specified in terms of paths of emissions (or emissions controls) over some relevant time horizon.

All of this means that dynamics matter. The chapter has shown how dynamic analysis – particularly the use of phase plane diagrams – can provide valuable insights into the analysis of pollution policy.

Further reading

Baumol and Oates (1988) is a classic source in this area, although the analysis is formal and quite difficult. Other useful treatments which complement the discussion in this chapter are Dasgupta (1982, chapter 8) and Smith (1972) which gives a very interesting mathematical presentation of the theory. Several excellent articles can be found in the volume edited by Bromley (1995).

The original references for stock pollution are Plourde (1972) and Forster (1975). Conrad and Olson (1992) apply this body of theory to one case, Aldicarb on Long Island. One of the first studies about the difficulties in designing optimal taxes (and still an excellent read) is Rose-Ackerman (1973). Pezzey (1996) surveys the economic literature on assimilative capacity, and an application can be found in Tahvonen (1995). Forster (1975) analyses a model of stock pollution in which the decay rate is variable.

Some journals provide regular applications of the economic theory of pollution. Of particular interest are the *Journal of Environmental Economics and Management*, *Ambio*, *Environmental and Resource Economics*, *Land Economics*, *Ecological Modelling*, *Marine Pollution Bulletin*, *Ecological Economics* and *Natural Resources Journal*.

Discussion question

1. In what principal ways do stock pollution models differ from models of flow pollutants?

Problem

1. Using equation 11.18, deduce the effect of an increase in α for a given value of r, all other things being equal, on:
   (a) M^* (b) A^*

CHAPTER 17 Renewable resources

It will appear, I hope, that most of the problems associated with the words 'conservation' or 'depletion' or 'overexploitation' in the fishery are, in reality, manifestations of the fact that the natural resources of the sea yield no economic rent. Fishery resources are unusual in the fact of their common-property nature; but they are not unique, and similar problems are encountered in other cases of common-property resource industries, such as petroleum production, hunting and trapping, etc.

Gordon (1954)

Learning objectives

After studying this chapter, the reader should be able to

- understand the biological growth function of a renewable resource, and the notions of compensation and depensation in growth processes
- interpret the simple logistic growth model, and some of its variants, including models with critical depensation
- understand the idea of a sustainable yield and the maximum sustainable yield
- distinguish between steady-state outcomes and dynamic adjustment processes that may (or may not) lead to a steady-state outcome
- specify, and solve for its bioeconomic equilibrium outcome, an open-access fishery, a static private-property fishery, and a present value (PV)-maximising fishery
- undertake comparative statics analysis and simple dynamic analysis for open-access and private-property models
- explain under what conditions the stock, effort and harvesting outcomes of private fisheries will not be socially efficient
- describe conditions which increase the likelihood of severe resource depletion or, in extreme cases, species extinction
- understand the workings, and relative advantages, of a variety of policy instruments that are designed to conserve renewable resource stocks and/or promote socially efficient harvesting

Introduction

Environmental resources are described as renewable when they have a capacity for reproduction and growth over human-relevant time frames. The class of renewable resources is diverse. It includes populations of biological organisms such as fisheries and forests which have a natural capacity for growth, and water and atmospheric systems which may be sustained (in quantitative and qualitative terms) by physical, chemical or biological processes. While the latter do not possess *biological* growth capacity, they do have some ability to assimilate pollution inputs (thereby maintaining their quality) and, at least in the case of water resources, can self-replenish as stocks are run down (thereby maintaining their quantity).

It is also conventional to classify arable and grazing lands as renewable resources. In these cases reproduction and growth take place by a combination of biological processes (such as the recycling of organic nutrients) and physical processes (irrigation, exposure to wind, etc.). Fertility levels can regenerate naturally so long as the demands made on the soil are not excessive. We may also consider more broadly defined environmental systems (such as wilderness areas or tropical moist forests) as being sets of interrelated renewable resources.

The categories just described are renewable *stock* resources. A broad concept of renewables would also include *flow* resources such as solar, wave, wind and geothermal energy. These share with biological stock resources the property that current harnessing of the flow does not mean that the total magnitude of the future flow will necessarily be smaller. Indeed, many forms of energy-flow resources are, for all practical purposes, non-depletable. The key point is that for renewable flow resources availability tomorrow is not affected at all by use today; this is not true of renewable stock resources.

Given this diversity of resource types, it will be necessary to restrict what will, and will not, be discussed here. Most of the literature on the economics of renewable resources is about two things: the harvesting of animal species ('hunting and fishing') and the economics of forestry. This chapter is largely concerned with the former; forestry economics is the subject of the following chapter. Agriculture could also be thought of as a branch of renewable resource harvesting. But agriculture – particularly in its more developed forms – differs fundamentally from other forms of renewable resource exploitation in that the environmental medium in which it takes place is designed and controlled. The growing medium is manipulated through the use of inputs such as fertilisers, pesticides, herbicides; temperatures may be controlled by the use of greenhouses and the like; and plant stocks are selected or even genetically modified. In that sense, there is little to differentiate a study of (developed) agricultural economics from the economics of manufacturing. For this reason, we do not survey the huge literature that is 'agricultural economics' in this text, although some of the environmental consequences of agricultural activity are discussed in the document *Agriculture.doc* available on the Companion Website. For reasons of space, we also do not cover the economics of renewable flow resources. A brief outline of some of the main issues in the economics of renewable energy resources is given in *Renewable Energy.doc*, again available on the Companion Website.

It is important to distinguish between stocks and flows of a renewable resource. The stock is a measure of the quantity of the resource existing at a point in time, measured either as the aggregate mass of the biological material in question (such as the total weight of fish of particular age classes or the cubic metres of standing timber), or in terms of population numbers. The flow is the change in the stock over an interval of time, where the change results either from biological factors, such as recruitment of new fish into the population through birth or exit due to natural death, or from harvesting activity.

While renewable stock resources and non-renewable resources differ fundamentally in terms of growth potential, both share the property that they are capable of being fully exhausted. That is, the stock can be driven to zero, sometimes irreversibly so, if excessive and prolonged harvesting or extraction activity is carried out. In the case of non-renewable resources, exhaustibility is a consequence of the finite nature of the stock. For renewable resources, although the stock can grow, it can also be driven to zero if conditions interfere with the reproductive capability of the renewable resource, or if rates of harvesting continually exceed net natural growth.

It is evident that enforceable property rights do not exist for many forms of renewable resource. In the absence of effective regulation or collective control over harvesting behaviour, the resource stocks are subject to open access. Open-access resources tend to be over-exploited in both a biological and an economic sense. As you will see, the likelihood of a renewable resource stock being harvested to the point of exhaustion is higher under open-access conditions than where enforceable property rights are established and access to harvesting can be controlled.

This chapter is principally about the harvesting of biological resources. Our exposition focuses on marine fishing. With some modifications, the fishery economics modelling framework can be used to analyse most kinds of renewable resource exploitation. The chapter is structured as follows. Section 17.1

sets out a simple model of biological growth that is widely used in modelling renewable resources and is particularly useful for studying commercial fisheries. Then, after a brief explanation of the concept of a 'steady-state' harvest, Sections 17.3 to 17.7 take the reader through the open-access fishery model. Our interest here lies in both equilibrium (or steady-state) outcomes (where they exist), and in the dynamic adjustment processes that take place as the fishery responds to exogenous changes in biological or economic parameters, or to shocks to the fishery.

Sections 17.8 and 17.9 consider a profit-maximising fishery in which enforceable property rights exist – what will be called here a 'private-property fishery'. Analysis proceeds in two stages. First we simplify by examining a 'static' model of the private-property fishery which abstracts from the passage of time. This is done by focusing attention only on steady-state (or equilibrium) outcomes in which variables are taken to be unchanging over time. The second stage generalises the fishing model to take account of the passage of time and the time value of money. Specifically, we set up an intertemporal model of a private-property fishery that is managed so as to maximise its present value over an infinite lifetime. All current-value flows are discounted at some positive rate to convert to present-value equivalents. Describing the second variant as a generalisation of the first is appropriate because – as we shall show – the static private fishery turns out to be a special case of the present-value-maximising fishery in which owners adopt a zero discount rate. Where discounting takes place at some positive rate, the outcomes of the two models differ. Section 17.10 demonstrates how the open-access, static private-property and present-value-maximising renewable stock resource models can be encompassing within a single analytical framework.

In common with the practice throughout this text, Section 17.10 examines the outcomes of the various commercial fishery regimes against the benchmark of a *socially efficient* fishery. We demonstrate that under a special set of conditions the harvesting programme of a competitive fishery where private property rights to the resource stocks are established and enforceable will be socially efficient. However, actual resource harvesting regimes are typically not socially efficient, even where attempts have been made to introduce private property rights. Among the reasons why they are not is the existence of various kinds of externalities. Open-access regimes will almost certainly generate inefficient outcomes, but externalities may also fail to be internalised in private-property fisheries. Fishery harvesting practices, as is the case with many other renewable resources, may also lead to outcomes that are considered undesirable on other grounds, particularly ones related to sustainability. The chapter concludes by examining a set of policy instruments that could be introduced in an attempt to move harvesting behaviour closer to that which is socially efficient or which is sustainable.

17.1 Biological growth processes

In order to investigate the economics of a renewable resource, it is first necessary to describe the pattern of biological (or other) growth of the resource. To fix ideas, we consider the growth function for a population of some species of fish. This is conventionally called a fishery. We suppose that this fishery has an intrinsic (or potential) growth rate denoted by g. This is the proportional rate at which the fish stock would grow when its size is small relative to the carrying capacity of the fishery, and so the fish face no significant environmental constraints on their reproduction and survival. The intrinsic growth rate g may be thought of as the difference between the population's birth and natural mortality rate (again, where the population size is small relative to carrying capacity). Suppose that the population stock is S and it grows at a fixed growth rate g. Then in the absence of human predation the rate of change of the population over time is given by[1]

[1] Be careful not to confuse a rate of change with a rate of growth. A rate of change refers to how much extra is produced in some interval of time. A rate of growth is that rate of change divided by its current size (to measure the change in proportionate terms). Note that we shall sometimes refer to an 'amount of growth' (as opposed to a growth rate); this should be read as the population size change over some interval of time.

$$\frac{\mathrm{d}S}{\mathrm{d}t} \equiv \dot{S} = gS \tag{17.1}$$

By integrating this equation, we obtain an expression for the stock level at any point in time:

$$S_t = S_0 \mathrm{e}^{gt}$$

in which S_0 is the initial stock level. In other words, for a positive value of g, the population grows exponentially over time at the rate g and without bounds. This is only plausible over a short span of time. Any population exists in a particular environmental milieu, with a finite carrying capacity, which sets bounds on the population's growth possibilities.

A simple way of representing this effect is by making the actual (as opposed to the potential) growth rate depend on the stock size. Then we have what is called density-dependent growth. Using the symbol χ to denote the actual growth rate, the growth function can then be written as

$$\dot{S} = \chi(S)S$$

where $\chi(S)$ states that χ is a function of S. Dividing both sides of this equation by S, we obtain

$$\frac{\dot{S}}{S} = \chi(S)$$

which shows explicitly the dependence of the actual growth rate on the stock size. If this function has the property that the proportionate growth rate of the stock ($\dot{S}/S$) declines as the stock size rises then the function is said to have the property of compensation.

Now let us suppose that under a given set of environmental conditions there is a finite upper bound on the size to which the population can grow (its carrying capacity). We will denote this as S_{MAX}. A commonly used functional form for $\chi(S)$ which has the properties of compensation and a maximum stock size is the simple logistic function:

$$\chi(S) = g\left(1 - \frac{S}{S_{\mathrm{MAX}}}\right)$$

in which the constant parameter $g > 0$ is what we have called the intrinsic or potential growth rate of the population. Where the logistic function determines the actual population growth rate, we may therefore write the biological growth function as

$$\dot{S} \equiv \frac{\mathrm{d}S}{\mathrm{d}t} = g\left(1 - \frac{S}{S_{\mathrm{MAX}}}\right)S \tag{17.2}$$

The changes taking place in the fish population that have been referred to so far are 'natural' changes. But as we want to use the notation $\dot{S}$ (or equivalently $\mathrm{d}S/\mathrm{d}t$) in the rest of this chapter to refer to the *net* effect of natural changes and human predation, we shall use the alternative symbol G to refer to stock changes due only to natural causes. (More completely, we shall use the notation $G(S)$ to make it clear that G depends on S.) With this change the logistic biological growth function is

$$G(S) = g\left(1 - \frac{S}{S_{\mathrm{MAX}}}\right)S \tag{17.3}$$

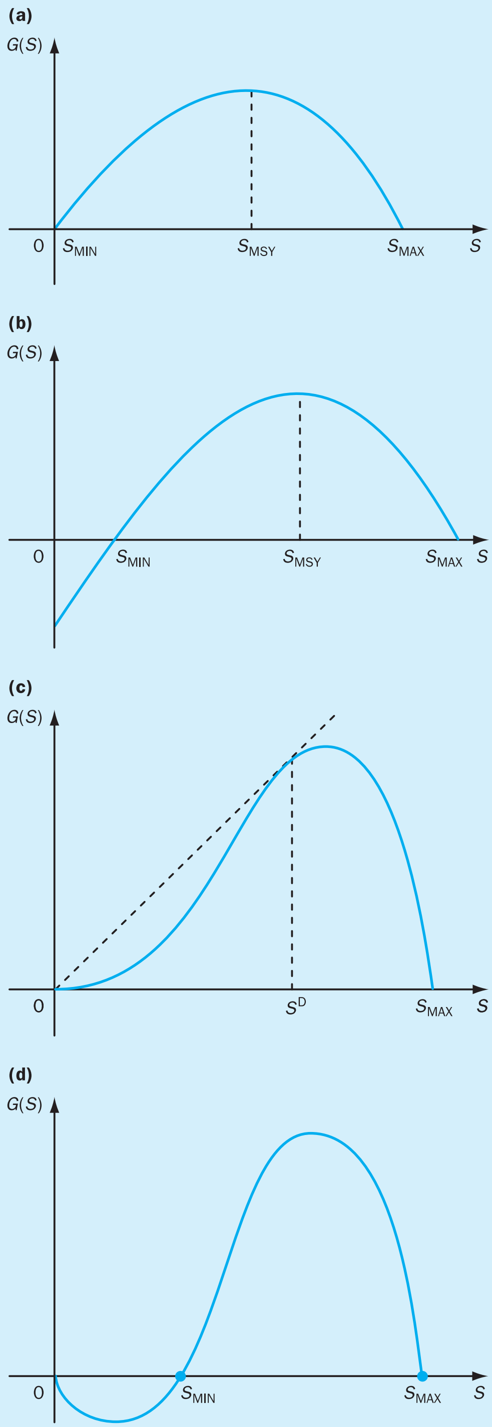
The logistic form is a good approximation to the natural growth processes of many fish, animal and bird populations (and indeed to some physical systems such as the quantity of fresh water in an underground reservoir). Some additional information on the simple logistic growth model, and alternative forms of logistic growth, is given in Box 17.1. Problem 1 at the end of the chapter investigates the logistic model a little further, and invites you to explore another commonly used equation for biological growth, the Gompertz function.

Box 17.1 The logistic form of the biological growth function

Logistic growth is one example of density-dependent growth: processes where the growth rate of a population depends on the population size. It was first applied to fisheries by Schaefer (1957). The equation for 'simple logistic growth' of a renewable resource population was given by equation 17.3.

Simple logistic growth is illustrated in Figure 17.1(a), which represents the relationship between the stock size and the associated rate of change of the population due to biological growth. Three properties should be noted by inspection of that diagram.

Box 17.1 *continued*



(a) S_{MAX} is the maximum stock size that can be supported in the environmental milieu. This value is conditional on the particular environmental circumstances that happen to prevail, and would change if any of those circumstances (such as ocean temperature or stocks of nutrients) change.
(b) By multiplication through of the terms on the right-hand side of equation 17.3, it is clear that the amount of growth, G, is a quadratic function of the resource stock size, S. For the simple logistic function, the maximum amount of growth (S_{MSY}) will occur when the stock size is equal to half of S_{MAX}.
(c) The amount of biological growth G is zero only at a stock size of zero and a stock size of S_{MAX}. For all intermediate values, growth is positive.

This last property may appear to be obviously true, but it turns out to be seriously in error in many cases. It implies that for any population size greater than zero natural growth will lead to a population increase if the system is left undisturbed. In other words, the population does not possess any positive lower threshold level.

However, suppose there is some positive population threshold level, S_{MIN}, such that the population would inevitably and permanently decline to zero if the actual population were ever to fall below that threshold. A simple generalisation of the logistic growth function that has this property is:

$$G(S) = g(S - S_{MIN})\left(1 - \frac{S}{S_{MAX}}\right) \tag{17.4}$$

Note that if $S_{MIN} = 0$, equation 17.4 collapses to the special case of equation 17.3. The generalisation given by equation 17.4 is illustrated in Figure 17.1(b). Several other generalisations of the logistic growth model exist. For example, the modified logistic model:

$$G(S) = gS^{\alpha}\left(1 - \frac{S}{S_{MAX}}\right)$$

Figure 17.1 (a) Simple logistic growth; (b) Logistic growth with a minimum population threshold; (c) Logistic growth with depensation; (d) Logistic growth with critical depensation

Box 17.1 ***continued***

has the property that for $\alpha > 1$ there is, at low stock levels, *depensation*, which is a situation where the proportionate growth rate $(G(S)/S)$ is an increasing function of the stock size, as opposed to being a decreasing function (compensation) in the simple logistic case where $\alpha = 1$. A biological growth function exhibiting depensation at stock levels below S^D (and compensation thereafter) is shown in Figure 17.1(c).

Finally, the generalised logistic function

$$G(S) = g\left(\frac{S}{S_{MIN}} - 1\right)\left(1 - \frac{S}{S_{MAX}}\right)S$$

exhibits what is known as *critical depensation*. As with equation 17.4, the stock falls irreversibly to zero if the stock ever falls below S_{MIN}. This function is represented in Figure 17.1(d). It should be evident that if a growth process does exhibit critical depensation, then the probability of the stock being harvested to complete depletion is increased, and increased considerably if S_{MIN} is a large proportion of S_{MAX}.

17.1.1 The status and role of logistic growth models

The logistic growth model is a stylised representation of the population dynamics of renewable resources. The model is most suited to non-migratory species at particular locations. Among fish species, demersal or bottom-feeding populations of fish such as cod and haddock are well characterised by this model. The populations of pelagic or surface-feeding fish, such as mackerel, are less well explained by the logistic function, as these species exhibit significant migratory behaviour. Logistic growth does not only fit biological growth processes. Brown and McGuire (1967) argue that the logistic growth model can also be used to represent the behaviour of a freshwater stock in an underground reservoir. However, a number of factors which influence actual growth patterns are ignored in this model, including the age structure of the resource (which is particularly important when considering stocks of long-lived species such as trees or whales) and random or chance influences. At best, therefore, it can only be a good approximation to the true population dynamics.

Judging the logistic model on whether it is the best available at representing any particular renewable resource would be inappropriate for our present purposes. One would not expect to find that biological or ecological modellers would use simple logistic growth functions. They will use more complex growth models designed specifically for particular species in particular contexts. But the needs of the environmental economist differ from those of ecological modellers. The former is willing to trade off some realism to gain simple, tractable models that are reasonably good approximations. It is for this reason that much economic analysis makes use of some version of the logistic growth function. Of more concern, perhaps, is the issue of whether it is appropriate to describe a biological growth process by any purely deterministic equation such as those given in Box 17.1. Ecological models typically specify growth as being stochastic, and linked in complex ways to various other processes taking place in more broadly defined ecosystems and subsystems. We shall briefly explore these matters later in the chapter.

17.2 Steady-state harvests

In this chapter, much of our attention will be devoted to steady-state harvests. Here we briefly explain the concept. Consider a period of time in which the amount of the stock being harvested (H) is equal to the amount of net natural growth of the resource (G). Suppose also that these magnitudes remain constant over a sequence of consecutive periods. We call this *steady-state* harvesting, and refer to the (constant) amount being harvested each period as a sustainable yield.

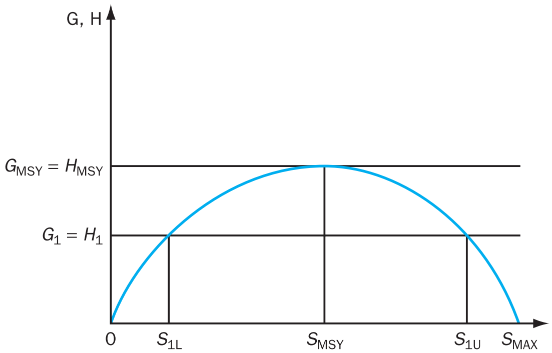


Figure 17.2 Steady-state harvests

Defining $\dot{S}$ as the actual rate of change of the renewable resource stock, with $\dot{S} = G - H$, it follows that in steady-state harvesting $\dot{S} = 0$ and so the resource stock remains constant over time. What kinds of steady states are feasible? To answer this, look at Figure 17.2. There is one particular stock size (S_{MSY}) at which the quantity of net natural growth is at a maximum (G_{MSY}). If at a stock of S_{MSY} harvest is set at the constant rate H_{MSY}, we obtain a *maximum sustainable yield* (MSY) steady state. A resource management programme could be devised which takes this MSY in perpetuity. It is sometimes thought to be self-evident that a fishery, forest or other renewable resource should be managed so as to produce its maximum sustainable yield. We shall see later that economic theory does not, in general, support this proposition.

H_{MSY} is not the only possible steady-state harvest. Indeed, Figure 17.2 shows that any harvest level represented by a point on the growth curve is a feasible steady-state harvest, and that any stock between zero and S_{MAX} can support steady-state harvesting. For example, H_1 is a feasible steady-state harvest if the stock size is maintained at either S_{1L} or S_{1U}. Which of these two stock sizes would be more appropriate for attaining a harvest level of H_1 is also a matter we shall investigate later.

Before moving on, it is important to understand that the concept of a steady state is a heuristic device: useful as a way of organising ideas and structuring analysis. But, like all heuristic devices, a steady state is a mental construct and using it uncritically can be inappropriate or misleading. Fisheries and other resource stocks are rarely, if ever, in steady states. Conditions are constantly changing, and the 'real world' is likely to be characterised by a more-or-less permanent state of disequilibrium. For some problems of renewable resource exploitation the analysis of transition processes is more important or insightful than information about steady states. We shall examine some of these 'dynamic' matters later in the chapter. Nevertheless, we will proceed on the assumption that looking at steady states *is* useful, and next investigate their properties under various institutional circumstances.

17.3 An open-access fishery

Our first model of renewable resource exploitation is an open-access fishery model. In conformity with the rest of this book, we study this using continuous-time notation. However, the equations which constitute the discrete-time equivalent of this continuous-time model (and others examined later in the chapter) are listed in full in Appendix 17.1. These will be used at various places in the chapter to give numerical illustrations of the arguments. The numerical values shown in our illustrative examples are computed using an Excel spreadsheet. Should the reader wish to verify that the values shown are correct, or to see how they would change under alternative assumptions, we have made the two spreadsheets used in our computations available to the reader on the Companion Web Site: *Comparative statics.xls* and *Fishery dynamics.xls*. While we hope that some readers (and instructors) will find these Excel documents useful, they are *not* necessary for an understanding of the contents of this chapter.

It is important to be clear about what an open-access fishery is taken to mean in the environmental economics literature. The open-access fishery model shares two of the characteristics of the standard perfect competition model. First, if the fishery is commercially exploited, it is assumed that this is done by a large number of independent fishing 'firms'. Therefore, each firm takes the market price of landed fish as given. Second, there are no impediments to entry into and exit from the fishery. But the free entry assumption has an additional implication

Table 17.1 The open-access fishery model

		General specification	Specific forms assumed
BIOLOGICAL SUB-MODEL:			
Biological growth	(17.5, 17.3)	$\mathrm{d}S/\mathrm{d}t = G(S)$	$G(S) = g\left(1 - \frac{S}{S_{\mathrm{MAX}}}\right)S$
ECONOMIC SUB-MODEL:			
Fishery production function	(17.6, 17.7)	$H = H(E, S)$	$H = eES$
Net growth of fish stock	(17.8)	$\mathrm{d}S/\mathrm{d}t = G(S) - H$	$\mathrm{d}S/\mathrm{d}t = g\left(1 - \frac{S}{S_{\mathrm{MAX}}}\right)S - H$
Fishery costs	(17.9, 17.10)	$C = C(E)$	$C = wE$
Fishery revenue	(17.11)	$B = PH$, P constant	$B = PH$, P constant
Fishery profit	(17.12)	$\mathrm{NB} = B - C$	$\mathrm{NB} = B - C = PeES - wE$
Fishing effort dynamics: open-access entry rule	(17.13)	$\mathrm{d}E/\mathrm{d}t = \delta \cdot \mathrm{NB}$	$\mathrm{d}E/\mathrm{d}t = \delta(PeES - wE)$
BIOECONOMIC EQUILIBRIUM CONDITIONS:			
Biological equilibrium	(17.14)	$G = H$	$G = H$
Economic equilibrium	(17.15)	$E = E^*$ at which $\mathrm{NB} = 0$	$E = E^*$ at which $\mathrm{NB} = 0$

Note: Numbers in parentheses refer to the appropriate equation number in the text.

in the open-access fishery, one which is *not* present in the standard perfect competition model.

In a conventional perfect competition model, each firm has enforceable property rights to its resources and to the fruits of its production and investment choices. However, in an open-access fishery, while owners have individual property rights to their fishing capital and to any fish that they have actually caught, they have no enforceable property rights to the *in situ* fishery resources, including the fish in the water.[2] On the contrary, any vessel is entitled or is able (or both) to fish wherever and whenever its owner likes. Moreover, if any boat operator chooses to leave some fish in the water in order that future stocks will grow, that owner has no enforceable rights to the fruits of that investment. It is as if a generalised 'what one finds one can keep' rule applies to fishery resources. We shall see in a moment what this state of affairs leads to. First, though, we need to set up the open-access fishery model algebraically.

The open-access model has two components:

1. a biological sub-model, describing the natural growth process of the fishery;
2. an economic sub-model, describing the economic behaviour of the fishing boat owners.

The model is laid out in full in Table 17.1. Subsequent parts of this section will take you through each of the elements described there. We shall be looking for two kinds of 'solutions' to the open-access model. The first is its equilibrium (or steady-state) solution. This consists of a set of circumstances in which the resource stock size is unchanging over time (a biological equilibrium) *and* the fishing fleet is constant with no net inflow or outflow of vessels (an economic equilibrium). Because the steady-state equilibrium is a joint biological–economic equilibrium, it is often referred to as *bioeconomic* equilibrium.

[2] This lack of *de facto* enforceability may derive from the property that the fish are spatially mobile. Or it may derive from the property that where fish stocks are spatially immobile no state, or any other entity, has jurisdiction, or does not or cannot exercise its jurisdiction.

The second kind of solution we shall be looking for is the adjustment path towards the equilibrium, or from one equilibrium to another as conditions change. In other words, our interest also lies in the dynamics of renewable resource harvesting. This turns out to have important implications for whether a fish population may be driven to exhaustion, and indeed whether the resource itself could become extinct. The properties of such adjustment paths are examined in Section 17.4.

17.3.1 The model described

17.3.1.1 Biological sub-model

In the absence of harvesting and other human interference, the rate of change of the stock depends on the prevailing stock size

$$dS/dt = G(S) \tag{17.5}$$

For our worked numerical example, we assume that the particular form taken by this growth function is the simple logistic growth model given by equation 17.3.

17.3.1.2 Economic sub-model

17.3.1.2.1 The harvest function (or fishery production function)

Many factors determine the size of the harvest, H, in any given period. Our model considers two of these. First, the harvest will depend on the amount of resources devoted to fishing. In the case of marine fishing, these include the number of boats deployed and their efficiency, the number of days when fishing is undertaken and so on. For simplicity, assume that all the different dimensions of harvesting activity can be aggregated into one magnitude called *effort*, E.

Second, except for schooling fisheries, it is probable that the harvest will depend on the size of the resource stock.[3] Other things being equal, the larger the stock the greater the harvest for any given level of effort. Hence, abstracting from other determinants of harvest size, including random influences, we may take harvest to depend upon the effort applied and the stock size. That is

$$H = H(E, S) \tag{17.6}$$

This relationship can take a variety of particular forms. One very simple form appears to be a good approximation to actual relationships (see Schaefer, 1954 and Munro, 1981, 1982), and is given by

$$H = eES \tag{17.7}$$

where e is a constant number, often called the catch coefficient.[4] Dividing each side by E, we have

$$\frac{H}{E} = eS$$

which says that the quantity harvested per unit effort is equal to some multiple (e) of the stock size. We have already defined the fish-stock growth function with human predation as the biological growth function less the quantity harvested. That is,

$$\dot{S} = G(S) - H \tag{17.8}$$

17.3.1.2.2 The costs, benefits and profits of fishing

The total cost of harvesting, C, depends on the amount of effort being expended

$$C = C(E) \tag{17.9}$$

For simplicity, harvesting costs are taken to be a linear function of effort,

$$C = wE \tag{17.10}$$

where w is the cost per unit of harvesting effort, taken to be a constant.[5]

[3] See Discussion Question 4 for more on this matter and the notion of schooling and non-schooling fisheries.

[4] The use of a constant catch coefficient parameter is a simplification that may be unreasonable, and is often dropped in more richly specified models. Note also that equation 17.7 can be regarded as a special case of the more general form $H = eE^{\alpha}S^{\beta}$ in which the exponents need not be equal to unity. In empirical modelling exercises, this more general form may be more appropriate. Another form of the harvest equation sometimes used is the exponential model $H = S(1 - \exp(-eE))$.

[5] The equation $C = wE$ imposes the assumption that harvesting costs are linearly related to fishing effort. However, Clark *et al.* (1979) explain that this assumption will be incorrect if capital costs are sunk (unrecoverable); moreover, they show that even in a private-property fishery (to be discussed later in the chapter), it can then be privately optimal to have severely depleted fish stocks as the fishery approaches its steady-state equilibrium (although the steady-state equilibrium itself is not affected by whether or not costs are sunk). We return to this matter later.

Let B denote the gross benefit from harvesting some quantity of fish. The gross benefit will depend on the quantity harvested, so we have

$$B = B(H)$$

In a commercial fishery, the appropriate measure of gross benefits is the total revenue that accrues to firms. Assuming that fish are sold in a competitive market, each firm takes the market price P as given and so the revenue obtained from a harvest H is given by[6]

$$B = PH \tag{17.11}$$

Fishing profit is given by

$$\mathrm{NB} = B - C \tag{17.12}$$

17.3.1.2.3 Entry into and exit from the fishery

To complete our description of the economic submodel, it is necessary to describe how fishing effort is determined under conditions of open access. A crucial role is played here by the level of economic profit prevailing in the fishery. Economic profit is the difference between the total revenue from the sale of harvested resources and the total cost incurred in resource harvesting. Given that there is no method of excluding incomers into the industry, nor is there any way in which existing firms can be prevented from changing their level of harvesting effort, effort applied will continue to increase as long as it is possible to earn positive economic profit.[7] Conversely, individuals or firms will leave the fishery if revenues are insufficient to cover the costs of fishing. A simple way of representing this algebraically is by means of the equation

$$\mathrm{d}E/\mathrm{d}t = \delta \cdot \mathrm{NB} \tag{17.13}$$

where δ is a positive parameter indicating the responsiveness of industry size to industry profitability. When economic profit (NB) is positive, firms will enter the industry; and when it is negative they will leave. The magnitude of that response, for any given level of profit or loss, will be determined by δ. Although the true nature of the relationship is unlikely to be of the simple, linear form in equation 17.13, this suffices to capture what is essential.

17.3.1.2.4 Bioeconomic equilibrium

We close our model with two equilibrium conditions that must be satisfied jointly. Biological equilibrium occurs where the resource stock is constant through time (that is, it is in a steady state). This requires that the amount being harvested equals the amount of net natural growth:

$$G = H \tag{17.14}$$

Economic equilibrium is only possible in open-access fisheries when rents have been driven to zero, so that there is no longer an incentive for entry into or exit from the industry, nor for the fishing effort on the part of existing fishermen to change. We express this by the equation

$$\mathrm{NB} = B - C = 0 \tag{17.15}$$

which implies (under our assumptions) that $PH = wE$. Notice that when this condition is satisfied, $\mathrm{d}E/\mathrm{d}t = 0$ and so effort is constant at its equilibrium (or steady-state) level $E = E^*$.

17.3.2 Open-access steady-state equilibrium

We can envisage an open-access fishery steady-state equilibrium by means of what is known as the fishery's yield–effort relationship. To obtain this, first note that in a biological equilibrium $H = G$. Then, by substituting the assumed functions for H and G from equations 17.7 and 17.3 respectively we obtain:

$$gS\left(1 - \frac{S}{S_{\mathrm{MAX}}}\right) = eES \tag{17.16}$$

[6] One could justify this assumption either by saying that the harvesting industry being examined is a small part of a larger overall market, or by arguing that the resource market is competitive, in which case each firm acts as if the market price is fixed (even though price will actually depend *ex post* on the realised total market supply).

[7] The terms rent, economic rent, royalties and net price are used as alternatives to economic profit. They all refer to a surplus of revenue over total costs, where costs include a proper allowance for the opportunity of capital tied up in the fishing fleet.

which can be rearranged to give

$$S = S_{MAX}\left(1 - \frac{e}{g}E\right) \tag{17.17}$$

Equation 17.17 is one equation in two endogenous variables, E and S (with parameters g, e and S_{MAX}). It implies a unique equilibrium stock at each level of effort.[8] Next substitute equation 17.17 into equation 17.7 ($H = eES$), giving

$$H = eES_{MAX}\left(1 - \frac{e}{g}E\right) \tag{17.18}$$

In an open-access economic equilibrium, profit is zero, so

$$PH = wE \tag{17.19}$$

Equations 17.18 and 17.19 constitute two equations in two unknowns (H and E); these can be solved for the equilibrium values of the two unknowns as functions of the parameters alone. The steady-state stock solution can then be obtained by substituting the expressions for H and E into equation 17.7. We list these steady-state solutions in Box 17.2, together with the numerical values of the steady-state levels of E, H and S under the particular set of assumptions about numerical values of the parameters given in Table 17.2.

This solution method can also be represented graphically, as shown in Figures 17.3 and 17.4. Figure 17.3 shows equilibrium relationships in stock–harvest space. The inverted U-shape curve is the logistic growth function for the resource. Three rays emanating from the origin portray the harvest–stock relationships (from the function $H = eES$) for three different levels of effort. If effort were at the constant level E_1, then the unique intersection of the harvest–stock relationship and biological growth function determines a steady-state harvest level H_1 at stock S_1. The lower effort level E_2 determines a second steady-state equilibrium (the pair $\{H_2, S_2\}$). We leave the reader to deduce why the label E_{MSY} has been attached to the third harvest–stock relationship. The various points of intersection satisfy equation 17.17, being equilibrium values of S for particular levels of E. Clearly there is an infinite quantity of possible equilibria, depending on what constant level of fishing effort is being applied.

The points of intersection in Figure 17.3 not only satisfy equation 17.17 but they also satisfy equation 17.18. Put another way, the equilibrium $\{E, S\}$ combinations also map into equilibrium $\{E, H\}$ combinations. The result of this mapping from $\{E, S\}$

Box 17.2 Analytical expressions for the open-access steady-state equilibrium and numerical solutions under baseline parameter assumptions

The analytical expressions for E^*, S^* and H^* (where an asterisk denotes the equilibrium value of the variable in question) as functions of the model parameters alone are:

$$E^* = \frac{g}{e}\left(1 - \frac{w}{PeS_{MAX}}\right) \tag{17.20}$$

$$S^* = \frac{w}{Pe} \tag{17.21}$$

$$H^* = \frac{gw}{Pe}\left(1 - \frac{w}{PeS_{MAX}}\right) \tag{17.22}$$

Derivations of expressions 17.20–17.22 are given in full in Appendix 17.2, available on the Companion Website. Throughout this chapter, we shall be illustrating our arguments with results drawn from fishery models using the parameter values shown in Table 17.2. At various points in the chapter we shall refer to these as the 'baseline' parameter values. It can be easily verified that for this set of parameter values, the steady-state solution is given by $E^* = 8.0$, $S^* = 0.2$ and $H^* = 0.024$.

Table 17.2 Parameter value assumptions for the illustrative numerical example

Parameter	Assumed numerical value
g	0.15
S_{MAX}	1
e	0.015
δ	0.4
P	200
w	0.6

[8] This uniqueness follows from the assumption that $G(S)$ is a simple logistic function; it may not be true for other biological growth models.

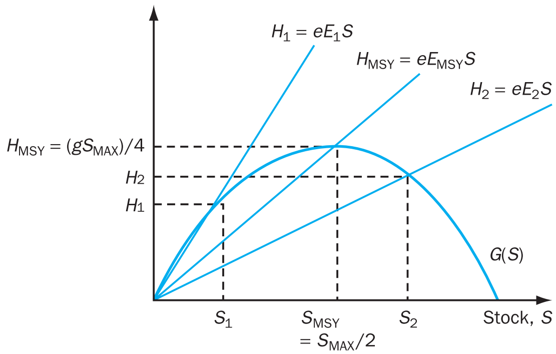


Figure 17.3 Steady-state equilibrium fish harvests and stocks at various effort levels

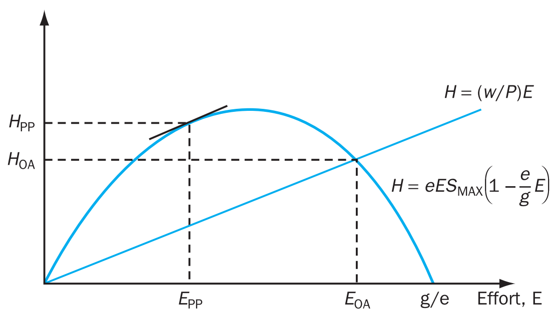


Figure 17.4 Steady-state equilibrium yield–effort relationship

space into $\{E, H\}$ space is shown in Figure 17.4. The inverted U-shape curve here portrays the steady-state harvests that correspond to each possible effort level. It describes what is often called the fishery's yield–effort relationship. Mathematically, it is a plot of equation 17.18.

The particular point on this yield–effort curve that corresponds to an open-access equilibrium will be the one that generates zero economic profit. How do we find this? The zero economic profit equilibrium condition $PH = wE$ can be written as $H = (w/P)E$. For given values of P and w, this plots as a ray from the origin with slope w/P in Figure 17.4. The intersection of this ray with the yield–effort curve locates the unique open-access equilibrium outcome.

Alternatively, multiplying both functions in Figure 17.4 by the market price of fish, P, we find that the intersection point corresponds to $PH = wE$. This is, of course, the zero profit condition, and confirms that $\{E_{OA}, H_{OA}\}$ is the open-access effort–yield equilibrium.

17.3.3 Comparative statics for the static open-access fishery model

It is instructive to inspect how the steady-state equilibrium magnitudes of effort, stock and harvest in an open-access fishery regime change in response to changes in the parameters or exogenous variables of that model. Equations 17.20, 17.21 and 17.22 (given in Box 17.2 in the previous section) give parametric equations for the open-access steady-state equilibrium values of effort, stock and harvest (denoted by E^*, S^* and H^* respectively) as functions of the model parameters alone are. For convenience, we reproduce these expressions below.

$$E^* = \frac{g}{e}\left(1 - \frac{w}{PeS_{MAX}}\right) \tag{17.20}$$

$$S^* = \frac{w}{Pe} \tag{17.21}$$

$$H^* = \frac{gw}{Pe}\left(1 - \frac{w}{PeS_{MAX}}\right) \tag{17.22}$$

Table 17.2 showed that for a particular illustrative (or baseline) set of parameter values, the steady-state equilibrium magnitudes of stock, effort and harvest were $S^* = 0.2$, $E^* = 8.0$ and $H^* = 0.024$ respectively. But we would also like to know how the steady-state equilibrium values would change in response to changes in the model parameters. This can be established using comparative static analysis.

Comparative static analysis is undertaken by obtaining the first-order partial derivatives of each of these three equations with respect to a parameter or exogenous variable of interest and inspecting the resulting partial derivative. The full set of partial derivatives is listed and derived in Appendix 17.2. What is of particular interest in comparative static analysis is to ascertain whether an unambiguous sign can be attached to a partial derivative, and if so to establish whether that sign is positive or negative. The magnitude of the derivative will also be of interest.

For example, how will S^* change as P or w changes? Inspection of the parametric expression for S^* shows that open-access S^* will increase if w increases and will decrease if P increases. More formally, the two partial derivatives are given by

$\partial S^*/\partial P = -w/P^2e$ and $\partial S^*/\partial w = 1/Pe$. As w, P and e are necessarily positive numbers, it follows that S^* will fall as P rises (or will rise as P falls) and that S^* will rise as w rises (or will fall as P falls). In these two cases, the sign of the effect is unambiguous. Sometimes the direction of an effect cannot be signed unambiguously; this should usually be evident by inspection of the partial derivative. Changes in the steady state value of H^*, for example, cannot be unambiguously signed with respect to changes in P, w or e, as can be seen from Table 17.3.

Table 17.3 lists the signs of a set of partial derivatives of interest. A plus sign means that the derivative is positive, a minus sign means that the derivative is negative, 0 means that the derivative is zero, and ? means that no sign can be unambiguously assigned to the derivative (and so we cannot say what the direction of the effect will be without knowing the actual values of the parameters that enter the partial derivative in question). So, for example, the – symbol in the upper left cell of the table indicates that the partial derivative of S^* with respect to P is negative; a change in P will be associated with a change in S^* in the opposite direction.

Note that variations in δ, the parameter indicating the responsiveness of industry size to industry profitability in an open-access regime, have no effect on any steady-state outcome (although, as we shall see shortly, they do affect how fast, if indeed at all, such an outcome may be achieved).

One should be aware, though, of the limitations of comparative static analysis. Take for example the parameter g, the intrinsic growth rate of the population of interest. Table 17.3 shows that the magnitude of the growth rate has no impact, other things remaining equal, on the steady-state equilibrium level of the resource stock. One might be tempted to infer from this that the likelihood of species extinction (or population collapse) is unaffected by how fast or slow the population grows. But that would be an inappropriate conclusion. The steady-state equilibrium value is unaffected; but the adjustment dynamics may be affected by the rate of g, and there is good reason to suppose that extinction is more likely for slow-growing species because of circumstances that arise along the dynamic adjustment paths that arise from parameter changes, or as a result of shocks or disturbances to the population of interest. It is to this matter that we now turn.

Table 17.3 Comparative static results: open access steady-state equilibrium

	P	w	e	g	δ
S^*	–	+	–	0	0
E^*	+	–	?	+	0
H^*	?	?	?	+	0

17.4 The dynamics of renewable resource harvesting

Our discussion so far has been exclusively about steady states: equilibrium outcomes which, if achieved, would remain unchanged provided that relevant economic or biological conditions remain constant. However, we may also be interested in the *dynamics* of resource harvesting. This would consider questions such as how a system would get to a steady state if it were not already in one, or whether getting to a steady state is even possible. In other words, dynamics is about transition or adjustment paths. Dynamic analysis might also give us information about how a fishery would respond, through time, to various kinds of shocks and disturbances.

A complete description of fishery dynamics is beyond the scope of this book. But some important insights can be obtained relatively simply. In this section, we undertake some dynamics analysis for the open-access model of Section 17.3. Table 17.1 laid out the equations of the open access model. Dynamic adjustment processes are driven by two differential equations, determining the instantaneous rates of change of S and E:

The net rate of change of the fish stock

$$\frac{\mathrm{d}S}{\mathrm{d}t} = G(S) - H = g\left(1 - \frac{S}{S_{\mathrm{MAX}}}\right)S - H \tag{17.8}$$

The open-access entry rule

$$\frac{\mathrm{d}E}{\mathrm{d}t} = \delta(PeES - wE) \tag{17.13}$$

When the fishery is in a steady-state equilibrium, by definition both $\mathrm{d}S/\mathrm{d}t$ and $\mathrm{d}E/\mathrm{d}t$ are zero. When the fishery is not in equilibrium the adjustment of S and E over time is governed by these two differential equations.[9] The properties of these dynamic adjustment processes can be explored by mathematical analysis.

To help fix ideas, suppose that a stock of fish exists which has not previously been commercially exploited. The stock size is, therefore, at its carrying capacity. This stock now becomes available for unregulated, open-access commercial exploitation. If the market price of fish, P, is reasonably high and fishing cost (per unit of effort), w, is reasonably low, the fact that stocks are high (and so easy to catch) implies that the fishery will be, at least initially, profitable for those who enter it. Have in mind equations 17.7, 17.10, 17.11 and 17.12 when thinking this through.

If a typical fishing boat can make positive economic profit then further entry will take place. How quickly new capacity is built up depends on the magnitude of the parameter δ in equation 17.13. In this early phase, effort is rising over time as new boats are attracted in, and stocks are falling. Stocks fall because harvesting is taking place while new recruitment to stocks is low: recall that the logistic growth function has the property that biological growth is near zero when the stock is near its maximum carrying capacity. This process of increasing E and decreasing S will persist for some time, but cannot last indefinitely. As stocks become lower, fish become harder to catch and so the cost per fish caught rises. Profits are squeezed from two directions: harvesting cost per fish rises, and fewer fish are caught.

Eventually, this profit squeeze will mean that a typical boat makes a loss rather than a profit, and so the process we have just described goes into reverse, with stocks rising and effort falling. In fact, for the model we are examining, the changes do not occur as discrete switches but instead are continuous and gradual. We find that stock and effort (and also harvest) levels have oscillatory cycles with the stock cycles slightly leading the effort cycles. These oscillations dampen down as time passes, and the system eventually settles to its steady-state equilibrium.

We illustrate such a transition process in Figures 17.5 and 17.6. These were generated using the Maple mathematical package, with parameter values as given in Table 17.2, and with initial values $E_0 = 0.1$ and $S_0 = 1$.[10] Figure 17.5 shows the convergent oscillatory behaviour of S and E as they move from their initial values to their steady-state equilibrium levels of $S^* = 0.2$ and $E^* = 8.0$. It is important to

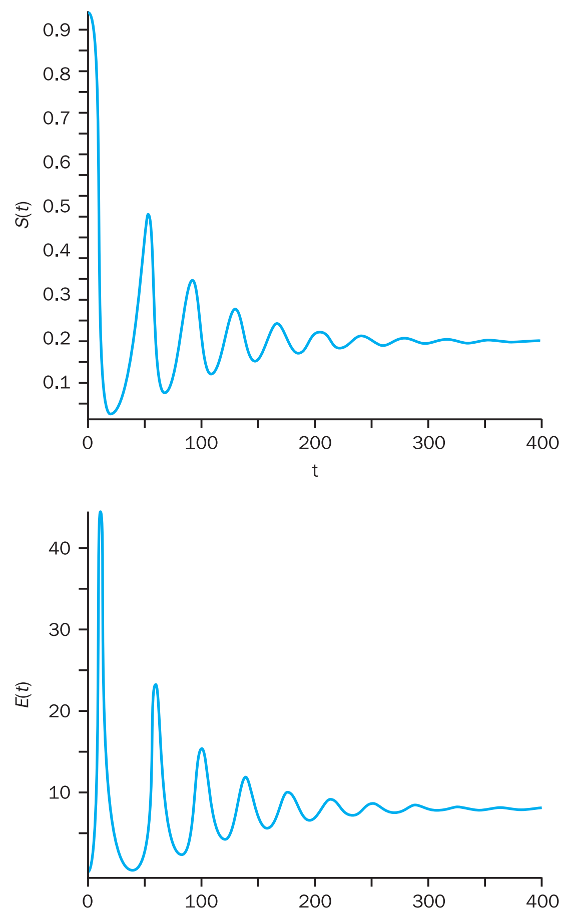


Figure 17.5 Stock and effort dynamic paths for the open-access model

[9] As $H = eES$, it follows that the two differential equations also determine the dynamic behaviour of the harvest rate, H.

[10] S_{MAX} has been normalised to unity (one) for convenience. So the fact that the initial value of S is 1 indicates that the fishery initially contains a stock at its environmental carrying capacity.

The Maple file implementing the simulations from which Figures 17.5 and 17.6 are taken (*Chapter17.mw*) is available from the Companion Website. For readers unable to read Maple files, an RTF file, readable via Word, is also available.

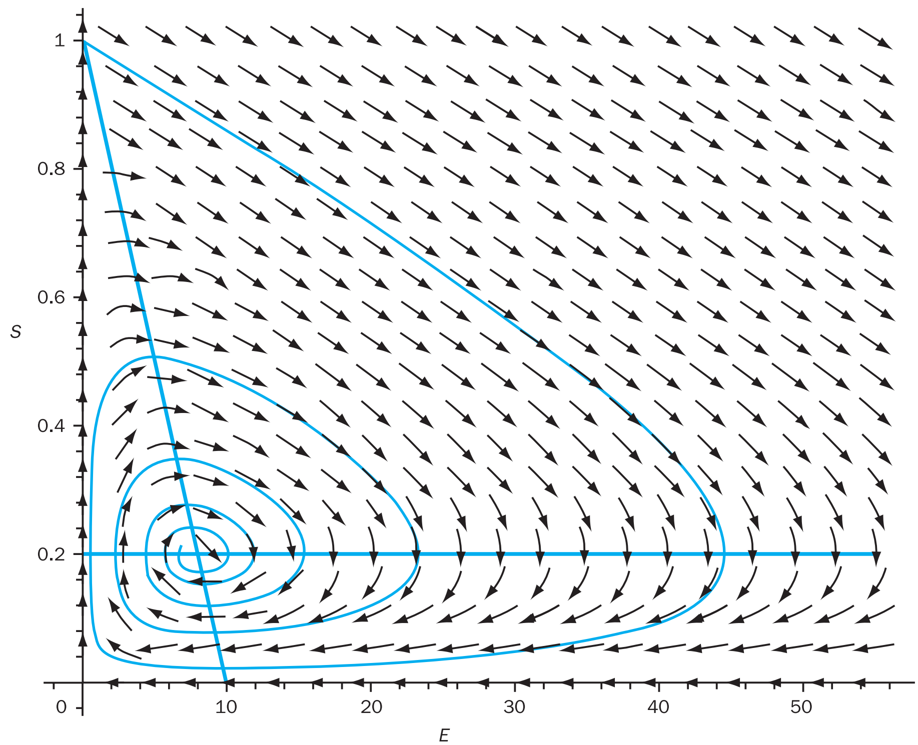


Figure 17.6 Phase-plane analysis of stock and effort dynamic paths for the illustrative model

understand that the frequency and amplitude of the oscillations in Figure 17.5 are obtained for one particular set of parameter values. For other combinations of parameter values, the cyclical behaviour of stock and effort will be different, and in some cases variability will be much less pronounced. Problem 7 at the end of this chapter invites you to use Maple or Excel (or both) to explore the properties of the open-access fishery model as values of parameters are varied.

The stable but oscillatory behaviour exhibited by the dynamic adjustment paths shown in Figure 17.5 – with E and S repeatedly over- and under-shooting equilibrium outcomes – is a characteristic feature of open-access fisheries. For the 'continuous time' specification of the open access model, the steady-state equilibrium outcomes implied by solving equations 17.20–22 will be achieved, albeit slowly. Furthermore, the continuous-time open-access model exhibits oscillatory but stable and convergent behaviour for any admissible sets of parameter values.[11]

Another way of describing the information given in Figure 17.5 is in terms of a 'phase-plane' diagram. An example is shown in Figure 17.6. The axes

[11] This is *not* true, however, in a discrete-time counterpart. If the model as specified in Table 17.1 is written in discrete-time form (using difference rather than differential equations for the changes in E and S over time), the resulting model dynamics can be entirely different! In particular, in discrete-time formulations, some combinations of parameters may induce oscillations so large as to drive the population down to a level from which it cannot recover, and so the fishery is driven out of existence. This can happen even though the parameters imply that a steady state exists with positive stock and effort. For example, in an Excel simulation explored in Problem 7 at the end of this chapter, a fish price of £270 rather than £200 (along with other parameter values as given in Table 17.2) suggests a steady-state solution of $S^* = 0.15$ and $E^* = 8.52$. (This can be verified by substituting parameter values in equations 17.21 and 17.20.) However, successive iterations of the discrete-time equations from initial values of $S = 1$ and $E = 1$ show that the stock collapses to zero after only 7 periods! (It does so because effort explodes so rapidly and to such a huge level that the stock is completely harvested in a short finite span of time.) This matter is explored a little further in Section 17.5.

define a space consisting of pairs of values for S and E. The steady-state equilibrium ($S^* = 0.2$ and $E^* = 8.0$) lies at the intersection of two straight lines the meaning of which will be explained in a moment. The adjustment path through time is shown by the line which converges through a series of diminishing cycles on the steady-state equilibrium. In the story we have just told, the stock is initially at 1 and effort begins at a small value (just larger than zero). Hence we begin at the top left point of the indicated adjustment path. As time passes, stock falls and effort rises – the adjustment path heads south-eastwards. After some time, the stock continues to fall but so too does effort – the adjustment path follows a south-west direction. Comparing Figures 17.5 and 17.6, you will see that the latter presents the same information as the former but in a different way.

But the phase-plane diagram also presents some additional information not shown in Figure 17.5. First, the arrows denote the directions of adjustment of S and E whenever the system is not in steady state, from any arbitrary starting point. It is evident that the continuous-time version of the open-access model we are examining here *in conjunction with* the particular baselines parameter values being assumed has strong stability properties: irrespective of where stock and effort happen to be the adjustment paths will lead to the unique steady-state equilibrium, albeit through a damped, oscillatory adjustment process.

Second, the phase-plane diagram explicitly shows the steady-state solution. As noted above this lies at the intersection of the two straight lines. The horizontal line is a locus of all economic equilibrium points at which profit per boat is zero, and so effort is unchanging ($\mathrm{d}E/\mathrm{d}t = 0$). The downward sloping straight line is a locus of all biological equilibrium points at which $G = H$, and so stock is unchanging ($\mathrm{d}S/\mathrm{d}t = 0$). Clearly, a bioeconomic equilibrium occurs at their intersection.[12]

17.5 Should one use a continuous-time model or a discrete-time model of the open-access fishery?

The open-access fishery model developed in this chapter (and listed in summary form in Table 17.1) is specified in terms of 'continuous time'. This is evident by inspecting equations 17.8 and 17.13. These are differential equations, describing the instantaneous rate of change of the net stock of fish and of fishing effort, respectively. However, one could easily rewrite the bioeconomic model in discrete-time terms, replacing equations 17.8 and 17.13 with their difference equation counterparts (giving amounts of change between successive discrete time periods). (See equation below.)

Which of these should one use? On first thinking about this, one might think the choice should be on the basis of analytical or computational convenience. For example, a reader familiar with spreadsheet software, such as Microsoft Excel, might find it more convenient to carry out model analysis in its discrete-time version; someone with fluency in a symbolic mathematical package such as Mathematica or Maple might have a preference for exploring a model in its continuous-time version.

Equation	**Continuous-time form**	**Discrete-time counterpart**
17.8	$\dfrac{\mathrm{d}S(t)}{\mathrm{d}t} = g\left(1 - \dfrac{S(t)}{S_{\mathrm{MAX}}}\right)S(t) - H(t)$	$S_{t+1} - S_t = g\left(1 - \dfrac{S_t}{S_{\mathrm{MAX}}}\right)S_t - H_t$
17.13	$\dfrac{\mathrm{d}E(t)}{\mathrm{d}t} = \delta(PeE(t)S(t) - wE(t))$	$E_{t+1} - E_t = \delta(PeE_tS_t - wE_t)$

[12] Figures 17.5 and 17.6 were generated using the Maple package to set up the continuous-time version of the open-access model. We provide, on the Companion Website, two files that readers may find useful. *Phase.doc* contains an elementary level account of phase-plane analysis, plus an explanation of how Maple – an easy-to-use and very powerful package – can generate the graphical output shown in Figure 17.6. The Maple file used to generate the picture is included as *Chapter 17.mw*, and a pdf version of that Maple file is also available.

If these two approaches yielded results that were exactly (or very closely) equivalent, then convenience would be an appropriate choice criterion. However, they do not. In some circumstances discrete-time modelling generates results very similar to those obtained from a continuous-time model. But this is not generally true. Indeed, the two approaches may differ not only quantitatively but also qualitatively. That is, discrete-time and continuous-time models of the same phenomenon will often yield results with very different properties.

This suggests that the choice should be made in terms of which approach best represents the phenomena being modelled. Let us first think about the biological component of a renewable resource model. Gurney and Nisbet (1998, p. 61) take a rather particular view on this matter and argue that

> The starting point for selecting the appropriate formalism must be therefore the recognition that real ecological processes operate in continuous time. Discrete-time models make some approximation to the outcome of these processes over a finite time interval, and should thus be interpreted with care. This caution is particularly important as difference equations are intuitively appealing and computationally simple. However, this simplicity is an illusion . . .
>
> . . . incautious empirical modelling with difference equations can have surprising (adverse) consequences.

It is undoubtedly the case that much ecological modelling is done in continuous time. But the Gurney and Nisbet implication that all ecological processes are properly modelled in continuous time would not be accepted by most ecologists. Flora and fauna where generations are discrete are always modelled in discrete time. Some ecologists, and others, would argue that discrete time is appropriate in other situations as well.

When it comes to the economic parts of bioeconomic models of renewable resources, it is also far from clear *a priori* whether continuous- or discrete-time modelling is more appropriate. Economists often have a predilection for continuous-time modelling, but that may often have a lot to do with physics emulation and mathematical convenience. Discrete-time modelling is appropriate for quite a range of economic problems; and when we get to the data, except for financial data, it's a discrete-time world. All that can really be said on this matter is that we should select between continuous- and discrete-time modelling according to the purposes of our analysis and the characteristics of the processes being modelled. For example, where one judges that a process being modelled involves discrete, step-like changes, and that capturing those changes is of importance to our research objectives, then the use of discrete-time difference equations would be appropriate.

For the reader interested in seeing how the Excel spreadsheet package can be used for dynamic simulation of the discrete-time counterpart of the open-access model used in this chapter, we have provided an annotated Excel workbook *Fishery dynamics.xls* on the Companion Website.[13] This Excel file implements the calculations used to obtain the discrete-time counterpart to Figures 17.5 and 17.6, carries out some additional dynamic simulations, and also illustrates other fishery models examined in this chapter. The workbook is set up so that the reader can easily carry out other simulations, and contains instructions for using the spreadsheet and an explanation of the simulation technique. A few minutes spent doing simulations in this discrete-time model counterpart will make it clear that dynamics are qualitatively very different from those shown in Figures 17.5 and 17.6.

17.6 Alternative forms of biological growth function in which there is a positive minimum viable population size

In this section and the one that follows we consider two extensions to the simple open-access model. This section deals with biological growth processes

[13] In essence, dynamic simulations are done in Excel in the following way. First choose initial values for E and S. Then, using the discrete-time versions of our model equations (listed in Appendix 17.1), calculate the values of H, E and S for the next time period. Next use the **Fill Down** function in Excel to copy the formulae which embody these model equations down the spreadsheet. In this way, Excel will recursively calculate levels of the variables over any chosen span of successive time periods. The steady-state solution can be found either analytically by using the parametric solution equations given in Box 17.2 or numerically by plotting successive values of H, E and S and observing at which numerical values the variables eventually settle down.

in which there is a positive minimum viable population size. Section 17.7 deals with the case in which fishery growth processes are stochastic.

Box 17.1 introduced several alternative forms of logistic growth. Of particular interest are those shown in panels (b) and (d) of Figure 17.1, in which there is a threshold stock (or population) size S_{MIN} such that if the stock were to fall below that level the stock would necessarily fall towards zero thereafter. There would be complete population collapse. In these circumstances, steady-state equilibria become irrelevant, as the fishery collapse prevents one from being obtained. Even though parameter values may suggest the presence of a steady-state equilibrium at positive stock, harvest and effort levels, if the stock were ever to fall below S_{MIN} during its dynamic adjustment process, the stock would irreversibly collapse to zero.

As a result, where a renewable resource has some positive threshold population size the likelihood of the stock of that resource collapsing to zero is higher than it is in the case of simple logistic growth, other things being equal. Moreover, the likelihood of stock collapse becomes greater the larger is S_{MIN}. These results are to a degree self-evident. A positive minimum viable population size implies that low populations that would otherwise recover through natural growth may collapse irretrievably to zero.

But matters are actually more subtle than this. In particular, where the renewable resource growth function exhibits critical depensation, the dynamics no longer exhibit the stable convergent properties shown in Figures 17.4 and 17.5. Indeed, over a large domain of the possible parameter configurations, the dynamics become unstable and divergent. In other words, the adjustment dynamics no longer take one to a steady-state equilibrium, but instead lead to population collapse. The stock can only be sustained over time by managing effort and harvest rates; but the management required for sustainability is, by definition, not possible in an open-access fishery. This illustrates two general points about renewable resources. First, under conditions of open access, the specific properties of the biological growth function for that resource have fundamental implications for its sustainability. Second, open-access conditions are likely to be incompatible with resource sustainability where biological growth is characterised by critical depensation.

17.7 Stochastic fishery models

To this point we have presumed that environmental conditions remain constant, or at least change in a deterministic and so predictable way over time. However, in practice the environment in which the resource is situated does change in an unpredictable way. When open access to a resource is accompanied by volatile environmental conditions, and where that volatility is fast-acting, unpredictable and with large variance, outcomes in terms of stock and/or harvesting can be catastrophic.

Simple intuition suggests why this is the case. Suppose that environmental stochasticity leads to there being a random component in the biological growth process. That is, the intrinsic growth rate contains a fixed component, g, plus a random component u, where the random component may be either positive or negative at some instant in time. Then the *achieved* growth in any period may be smaller or larger than its underlying mean value as determined by the growth equation 17.3. When the stochastic component happens to be negative, and is of sufficient magnitude to dominate the positive fixed component, actual growth is negative. If the population were at a low level already, then a random change inducing lower than normal, or negative, net growth could push the resource stock below the point from which biological recovery is possible. This will, of course, be more likely the further above zero is S_{MIN}.

17.8 The private-property fishery

In an open-access fishery, firms exploit available stocks as long as positive profit is available. While this condition persists, each fishing vessel has an incentive to maximise its catch. But there is a dilemma here, both for the individual fishermen and for society as a whole. From the perspective of the fishermen, the fishery is perceived as being overfished. Despite each boat owner pursuing maximum profit, the collective efforts of all drive profits down to zero. From a social perspective, the fishery will be economically 'overfished' (this is a result that has not yet been established, but it will be later, in Section 17.11). Moreover, the stock level may be driven down to levels that are considered to be dangerous on biological or sustainability grounds.

What is the underlying cause of this state of affairs? Although reducing the total catch today may be in the collective interest of all (by allowing fish stocks to recover and grow), it is not rational for any fisherman to individually restrict fishing effort. There can be no guarantee that he or she will receive any of the rewards that this may generate in terms of higher catches later. Indeed, there may not be any stock available in the future. In such circumstances, each firm will exploit the fishery today to its maximum potential, subject only to the constraint that its revenues must at least cover its costs.

A particular set of institutional arrangements could overcome some of these dilemmas: the 'private-property' fishery. This kind of fishery – and several variants of it – has been explored in the fishery economics literature. However, discussions of the private-property fishery rarely make explicit the institutional assumptions that lie behind it. It is important to do so. What is described in this book as a private-property fishery has the following three characteristics:

1. There are a large number of fishing firms, each behaving as a price-taker and so regarding price as being equal to marginal revenue. It is for this reason that the industry is often described as being competitive.
2. Each firm is wealth maximising.
3. There is a particular structure of well-defined and enforceable property rights to the fishery, such that owners can control access to the fishery and appropriate any rents that it is capable of delivering.

What exactly is this particular structure of private property rights? Within the literature there are several (sometimes implicit) answers to this question. We shall outline two of them. In one view, a private-property fishery refers to the harvesting of a single species from lots of different biological fisheries, fishing grounds. Each fishing ground is owned by a fishing firm. That fishing firm has private-property rights to the fish which are on that fishing ground currently and at all points in time in the future.[14] All harvested fish, however, sell in one aggregate market at a single market price. A second view regards the fishery as being managed by a single entity which controls access to the fishery and coordinates the activity of individual operators to maximise total fishery profits (or wealth). Nevertheless, harvesting and pricing behaviour are competitive rather than monopolistic.

Neither of these accounts is descriptively accurate in terms of what currently exists or in terms of what might realistically exist. The first faces problems in deciding how to specify ownership rights to migratory fish. Moreover, it could only be descriptively accurate if the fishery in question is a huge, highly spatially aggregated, fishery. The researcher does not usually want to study at this level of aggregation. The second concept – the coordinated fishery – seems problematic in that we rarely, if ever, find examples of such *internally* coordinated fisheries (except in the case of fish farming and the like). And even if one were to find examples, it is difficult to imagine that they would operate as competitive fisheries rather than as monopolies or cartels.[15]

But to label one or both of these views as descriptively unrealistic is to miss the point that we wish to think of them in 'as if' terms. That is, we want a specification such that the industry behaves *as if* each firm has its own 'patch' of fishery that others are not permitted to exploit or *as if* it were coordinated in the way mentioned above. Given either of these *as if* assumptions, one can then reasonably assume that owners undertake economically rational management decisions, and are in a position to make

[14] The owners of any fishing firm may lease or sell their property rights to another set of individuals.

[15] In fact, two other variants of the private-property fishery sometimes discussed in the literature are actually these: the monopoly fishery and the cartel fishery. However, given the fact that they are so uncommon in practice, we do not deal with those models in this text, except for a brief reference to a monopoly fishery in Section 17.11.4 and also in Appendix 17.3.

investment decisions confident in the belief that the returns on any investment made can be individually appropriated. This is what distinguishes a private-property fishery fundamentally from an open-access fishery.

An important benefit from thinking about property rights in this way is the pointers it gives in developing public policy towards fishery regulation and management. If we are confident that a particular property rights structure would bring about socially efficient (or otherwise desirable) outcomes, then policy instruments can be designed to mimic that structure. We will argue below that an individual transferable quota (ITQ) fishing permit system can be thought of in this way.

17.8.1 The static profit-maximising private-property fishery model

As we explained in the Introduction to this chapter, our analysis of the private-property fishery proceeds in two steps. The first, covered in this section, develops a simple static model of a private-property fishery in which the passage of time is not explicitly dealt with. In effect, the analysis supposes that biological and economic conditions remain constant over some span of time. It then investigates what aggregate level of effort, stock and harvest would result if each individual owner (with enforceable property rights) managed affairs so as to maximise profits over any arbitrarily chosen interval of time. This way of dealing with time – in effect, abstracting from it, and looking at decisions in only one time period but which are replicated over successive periods – leads to its description as a *static* fishery model. We shall demonstrate later that the static private-property fishery turns out to be a special case of a multi-period fishery model: the special case in which owners use a zero discount rate.

The biological and economic equations of the static private-property fishery model are identical to those of the open-access fishery in all respects but one: the open-access entry rule ($dE/dt = \delta \cdot NB$), which in turn implies a zero-profit economic equilibrium, no longer applies. Instead, owners choose effort to maximise economic profit from the fishery. This can be visualised by looking back to Figure 17.4. Multiply both functions by the market price of fish. The inverted U-shape yield–effort equation then becomes a revenue–effort equation. And the ray emerging from the origin now becomes $PH = wE$, with the right-hand side thereby denoting fishing costs. Profit is maximised at the effort level which maximises the surplus of revenue over costs. Diagrammatically, this occurs where the slopes of the total cost and total revenue curves are equal. This is indicated in Figure 17.4 by the tangent to the yield–effort function at E_{PP} being parallel to the slope of the $H = (w/P)E$ line.

An algebraic derivation of the steady-state solution to this problem – showing stock, effort and harvest as functions of the parameters – is given in Box 17.3. It is easy to verify from the solution equations given there that the steady-state values of E, S and H are given by $E^*_{PP} = 4.0$, $S^*_{PP} = 0.60$ and $H^*_{PP} = 0.0360$. To facilitate comparison, parametric solution equations for the steady-state equilibrium stock, harvest and effort, and their numerical values under baseline parameter assumptions, for both open-access and static private-property fishery, are reproduced in Table 17.4.

Table 17.4 Steady-state open-access and static private-property equilibria compared: parametric solution equations and numerical values of steady state solutions under baseline parameter value assumptions

	Open access		Static private property	
	Open access parametric solution equation	Baseline numerical value	Static private property parametric solution equation	Baseline numerical value
Stock	$S^* = \dfrac{w}{Pe}$	0.200	$S^*_{PP} = \dfrac{1}{2}\dfrac{PeS_{MAX} + w}{Pe}$	0.600
Effort	$E^* = \dfrac{g}{e}\left(1 - \dfrac{w}{PeS_{MAX}}\right)$	8.000	$E^*_{PP} = \dfrac{1}{2}\dfrac{g}{e}\left(1 - \dfrac{w}{PeS_{MAX}}\right)$	4.000
Harvest	$H^* = \dfrac{gw}{Pe}\left(1 - \dfrac{w}{PeS_{MAX}}\right)$	0.024	$H^*_{PP} = \dfrac{1}{4}g\left(S_{MAX} - \dfrac{w^2}{P^2e^2S_{MAX}}\right)$	0.036

Box 17.3 Derivation of the static private-property steady-state equilibrium for our assumed functional forms

The derivation initially follows exactly that given in Section 17.3.2, with equations 17.16 to 17.18 remaining valid here. However, the zero profit condition (equation 17.19) is no longer valid, being replaced by the profit-maximisation condition:

$$\text{Maximise NB} = PH - wE \tag{17.23}$$

Remembering that $H = eES$, and treating E as the instrument variable, this yields the necessary first-order condition,

$$\partial(PeES)/\partial E = \partial(wE)/\partial E \tag{17.24}$$

Substituting equation 17.17 into 17.24 we have

$$\partial\left(PeES_{\text{MAX}}\left(1 - \frac{e}{g}E\right)\right)/\partial E = \partial(wE)/\partial E$$

from which we obtain after differentiation

$$PeS_{\text{MAX}} - 2PES_{\text{MAX}}\left(\frac{e^2}{g}\right) = w \tag{17.25}$$

That is, the marginal revenue of effort is equal to the marginal cost of effort. This can be solved for E^*_{PP} (the subscript denoting 'private property') to give

$$E^*_{\text{PP}} = \frac{1}{2}\frac{g}{e}\left(1 - \frac{w}{PeS_{\text{MAX}}}\right) \tag{17.26}$$

Substitution of E^*_{PP} into 17.17 gives

$$S^*_{\text{PP}} = \frac{1}{2}\frac{PeS_{\text{MAX}} + w}{Pe} \tag{17.27}$$

and then using $H = eES$ we obtain[16]

$$H^*_{\text{PP}} = \frac{1}{4}g\left(S_{\text{MAX}} - \frac{w^2}{P^2e^2S_{\text{MAX}}}\right) \tag{17.28}$$

[16] In the Excel spreadsheets available on the Companion Website, an alternative (but exactly equivalent) version of this expression has been used to generate the Excel formulas, namely

$$H^*_{\text{PP}} = \frac{1}{4}\frac{g(PeS_{\text{MAX}} - w)(PeS_{\text{MAX}} + w)}{P^2e^2S_{\text{MAX}}}$$

Under our assumptions regarding functional forms, the static private-property equilibrium will always lead to a higher resource stock level and a lower effort level than that which prevails under open access. This is confirmed for our particular parameter assumptions, with the private-property stock being three times higher and effort only half as large as in open access.

The steady-state harvest may be higher, lower or identical. This is evident from inspection of Figure 17.4. For the particular set of parameter values used in the illustrative example, private-property harvest is *larger* than open-access harvest, as shown in the diagram. But it will not always be true that private-property harvests exceed those under open access. For example, if P were 80 (rather than 200) and all other parameter values were those specified in the baseline set listed in Table 17.2 then an open-access fishery would produce $H = 0.0375$, the maximum sustainable yield of the fishery. In contrast, a private-property fishery would in those circumstances yield only $H = 0.0281$.

The source of this indeterminacy follows from the inverted U shape of the yield–effort relationship. Although stocks will be higher under private property than open access, the quadratic form of the stock–harvest relationship implies that harvests will not necessarily be higher with higher stocks.

17.8.2 Comparative statics

For convenience, we list in Table 17.4 the expressions obtained in earlier sections for the steady-state equilibria of E, H and S under both open access and static private property regimes. As discussed earlier in Section 17.3.3, we can use these expressions to carry out comparative static analysis: that is, to make qualitative predictions about the effects of changing a particular parameter on the equilibrium levels of those three variables. The results are listed in Table 17.5.

It is evident from Table 17.5 is that, in terms of the signs of the appropriate partial derivatives, open access and static private property are identical for the steady-state equilibrium stock and effort.[17] However,

[17] Be careful here. The signs of the partial derivatives are identical, but their mathematical expressions and numerical values are not.

Table 17.5 Comparative static results: open access compared with static private property

	P	w	E	g	δ
Open access					
S^*	−	+	−	0	0
E^*	+	−	?	+	0
H^*	?	?	?	+	0
Static private property					
S^*_{PP}	−	+	−	0	0
E^*_{PP}	+	−	?	+	0
H^*_{PP}	+	−	+	+	0

for the steady-state harvest level, three partial derivatives that cannot be signed under open-access conditions show unambiguous directions of response in the (static) private property case.[18]

17.8.3 The present-value-maximising fishery model

The present-value-maximising fishery model generalises the model of the static private-property fishery, and in doing so provides us with a sounder theoretical basis and a richer set of results. The essence of this model is that a rational private-property fishery will organise its harvesting activity so as to maximise the discounted present value (PV) of the fishery. This section specifies the present-value-maximising fishery model and describes and interprets its main results. Full derivations have been placed in Appendix 17.3. The individual components of our model are very similar to those of the static private fishery model. However, we now bring time explicitly into the analysis by using an intertemporal optimisation framework. Initially we shall develop results using general functional forms. Later in this section, solutions are obtained for the specific functions and baseline parameter values assumed earlier in this chapter.

As in previous sections of the chapter, it is assumed that the market price of fish is a constant, exogenously given, number, and that the market for landed fish is competitive.[19] It will be convenient to regard harvest levels as the instrument (control) variable. To facilitate this, we specify fishing costs as a function of the quantity harvested and the size of the fish stock. Moreover, it is assumed that costs depend positively on the amount harvested and negatively on the size of the stock.[20,21] That is,

$$C_t = C(H_t, S_t)\ C_H > 0,\ C_S < 0$$

The initial population of fish is S_0, the natural growth of which is determined by the function $G(S)$. The fishery owners select a harvest rate for each period over the relevant time horizon (here taken to be infinity) to maximise the present value (or wealth) of the fishery, given an interest rate i. Algebraically, we express this as

$$\text{Max} \int_0^{\infty} \{PH_t - C(H_t, S_t)\}e^{-it}\text{d}t$$

subject to

$$\frac{\text{d}S}{\text{d}t} = G(S_t) - H_t$$

and initial stock level $S(0) = S_0$.[22] The necessary conditions for maximum wealth include

[18] See Problem 7 for a further examination of this matter.

[19] However, Appendix 17.3 will also go through the more general case in which the market price of fish varies with the size of total industry catch, and will briefly examine a monopolistic fishery.

[20] The reader may be confused about our formulation of the harvest cost function. In an earlier section, we wrote $C = C(E)$, equation 17.9. But note that we have also assumed that $H = H(E, S)$, equation 17.6. If 17.6 is written as E in terms of H and S, and that expression is then substituted into 17.9, we obtain $C = C(H, S)$. It is largely a matter of convenience whether we express costs in terms of effort or in terms of harvest and stock. In our discussion of open-access equilibrium, we chose to regard fishing effort as a variable of interest and did not make that substitution. In this section, our interest lies more in the variable H and so it is convenient to make the substitution. But the results of either approach can be found from the other.

[21] There is another issue here that we should mention. The costs of fishing should include a proper allowance for all the opportunity costs involved. For land-based resources, the land itself is likely to have alternative uses, and so its use in any one activity will have a land opportunity cost. For fisheries, however, there is rarely an alternative commercial use of the oceans, and so this kind of opportunity cost is not relevant. However, from a social point of view there may be important alternative uses of the oceans (for example, as conserved sources of biodiversity). Hence a difference can exist between costs as seen from a social and a private point of view.

[22] As we are now dealing explicitly with the passage of time, it is necessary once again to introduce appropriate time notation into the equations of the model. The model continues to be specified in terms of continuous time. However, for compactness of notation, we use X_t to denote the value of X at the instant of time, t, rather than the more common notation $X(t)$.

$$p_t = P - \frac{\partial C(H, S)}{\partial H_t} \quad (17.29)$$

$$\frac{dp_t}{dt} = ip_t - p_t \frac{dG(S)}{dS_t} + \frac{\partial C(H, S)}{\partial S_t} \quad (17.30)$$

It is very important to distinguish between upper-case P and lower-case p in equation 17.29, and in many of the equations that follow. P is the market, or landed, or gross price of fish; it is, therefore, an observable quantity. As the market price is being treated here as an exogenously given fixed number, no time subscript is required on P. In contrast, lower-case p_t is a shadow price, which measures the contribution to wealth made by an additional unit of fish stock at the wealth-maximising optimum. It is also known as the net price of fish, and also as unit rent. (Note it is p rather than P which denotes the resource net price.)

Three properties of the net price deserve mention. First, it is typically an unobservable quantity. Second, like all shadow prices, it will vary over time, unless the fishery is in steady-state equilibrium. In general, therefore, it is necessary to attach a time label to net price.

The third property concerns an equivalence between the net price equations for the renewable and non-renewable resource cases. Equation 17.29 defines the net price of the resource (p_t) as the difference between the market price and marginal cost of an incremental unit of harvested fish. Differential equation 17.30 governs the behaviour over time of the net price, and implicitly determines the harvest rate in each period, as we shall see in the discussions below. Note that if growth is set to zero (as would be the case for a non-renewable resource) and the stock term in the cost function is not present (so that the size of the resource stock has no impact on harvest costs) then the net price equation for renewable resources (equation 17.30 above) collapses to a special case that is identical to its counterpart for a non-renewable resource (see equation 14.6 in Chapter 14 or the expressions for net price in section 15.3 in Chapter 15). Thus the differential equation for the net price of a non-renewable resource turns out to be a special case of a more general differential equation expression for the net price of a renewable resource (a result which we also noted earlier in Chapter 14).

17.8.3.1 Steady-state equilibrium in the present-value-maximising fishery

We first investigate the properties of a PV-maximising fishery in steady-state equilibrium. In a steady state all variables are unchanging with respect to time, which implies that $dp/dt = 0$ and also that $G(S) = H$. Hence the optimising conditions 17.29 and 17.30 collapse to the simpler forms

$$p = P - \frac{\partial C(H, S)}{\partial H} \quad (17.31)$$

$$ip = p\frac{dG(S)}{dS} - \frac{\partial C(H, S)}{\partial S} \quad (17.32)$$

17.8.3.2 Interpretation of equation 17.32

How is the present value of profits maximised? The key to understanding profit-maximising behaviour when access to the resource can be regulated lies in capital theory. A renewable resource is a capital asset. To fix ideas, think about a fishery with a single owner who can control access to the resource and appropriate all returns from it. We wish to consider the owner's decision about whether to marginally change the amount of fish harvesting currently being undertaken. Specifically, our interest lies in the marginal cost and the marginal benefit of changing the harvest rate for one period only.

A decision about whether to defer some harvesting until the next period is made by comparing the marginal costs and benefits of adding additional units to the resource stock. What is the marginal cost of deferring the harvesting of one unit of fish from the current period until the following period? Choosing not to harvest now is equivalent to a capital investment. The uncaught fish will be there next period; moreover, biological growth will mean that there is an additional increment to the stock next period, over and above the quantity of fish left unharvested. This amounts to saying that the asset – in this case the fishery – is productive. By not harvesting an incremental unit this period, the fisher incurs an opportunity cost that consists of the forgone return by holding a stock of unharvested fish. The marginal cost of the investment is ip, as the sale of one unit of the harvested fish would have led to a revenue net of harvesting costs (given by the net price of the resource, p) that could have earned the prevailing rate of return on capital, i. Since we are considering a decision to defer this revenue by one period, the present value of this sacrificed return is ip.

The owner compares this marginal cost with the marginal benefit obtained by not harvesting the incremental unit this period. There are two categories of benefit:

1. As an additional unit of stock is being added, *total* harvesting costs will be reduced by the quantity $C_S = \partial C/\partial S$ (note that $\partial C/\partial S < 0$).
2. The additional unit of stock will grow by the amount dG/dS. The value of this additional growth is the amount of growth valued at the net price of the resource.

A present-value-maximising owner will add units of resource to the stock provided the marginal cost of doing so is less than the marginal benefit. That is:

$$ip < \frac{dG}{dS}p - \frac{\partial C}{\partial S}$$

This states that a unit will be added to stock provided its 'holding cost' (ip) is less than the sum of its harvesting cost reduction and value-of-growth benefits. Conversely, a present-value-maximising owner will harvest additional units of the stock if marginal costs exceed marginal benefits:

$$ip > \frac{dG}{dS}p - \frac{\partial C}{\partial S}$$

These imply the asset equilibrium condition, equation 17.32. When this is satisfied, the rate of return the resource owner obtains from the fishery is equal to i, the rate of return that could be obtained by investment elsewhere in the economy. This is one of the intertemporal efficiency conditions we identified in Chapter 11. To confirm that this equality exists, divide both sides of equation 17.32 by the net price p to give

$$i = \frac{dG}{dS} - \frac{\left(\frac{\partial C}{\partial S}\right)}{p} \tag{17.33}$$

Equation 17.33 is a version of what is sometimes called the 'fundamental equation' of renewable resources. The left-hand side is the rate of return that can be obtained by investing in assets elsewhere in the economy. The right-hand side is the rate of return that is obtained from the renewable resource. This is made up of two elements:

- the natural rate of growth in the stock from a marginal change in the stock size;
- the value of the reduction in harvesting costs that arises from a marginal increase in the resource stock.

Equation 17.33 is an implicit equation for the unknown PV-maximising equilibrium value of S. Solving it for S gives an analytical expression for the equilibrium stock.[23] Given that, expressions for the PV-optimal solutions for H and E can be obtained using the biological growth function and the fishery production function, equation 17.6. Table 17.6 lists the equilibrium values of S, E and H for interest rates between zero and 100% (and two higher values), conditional on our assumed functional forms and other baseline parameter values. For purposes of comparison we also show the equilibrium values for a static private-property fishery and an open-access fishery.[24] Note that for these last two models, equilibrium values do not vary with the interest rate. Three important results are evident from an inspection of the data in Table 17.6:

1. In the special case where the interest rate is zero, the steady-state equilibria of a static private-property fishery and a PV-maximising fishery are identical. For all other interest rates they differ.
2. For non-zero interest rates, the steady-state fish stock (fishing effort) in the PV profit-maximising fishery is lower (higher) than that in the static private fishery, and becomes increasingly lower (higher) the higher is the interest rate.
3. As the interest rate becomes arbitrarily large, the PV-maximising outcome converges to that of an open-access fishery.

23 The solution is derived in Appendix 17.3.

24 These calculations are implemented in both the Excel spreadsheet *Comparative statics.xls* and in the Maple file *Chapter 17.mw*.

Table 17.6 Steady-state equilibrium outcomes from the illustrative spreadsheet model for baseline parameter values in (a) a static private-property fishery, (b) a PV-maximising fishery with various interest rates and (c) an open-access fishery

	Static private fishery*			PV-maximising fishery			Open-access fishery*		
i	S	E	H	S	E	H	S	E	H
0.0	0.6	4.0	0.036	0.6000	4.0000	0.0360	0.20	8.0	0.024
0.1	0.6	4.0	0.036	0.4239	5.7607	0.0366	0.20	8.0	0.024
0.2	0.6	4.0	0.036	0.3333	6.6667	0.0333	0.20	8.0	0.024
0.3	0.6	4.0	0.036	0.2899	7.1010	0.0309	0.20	8.0	0.024
0.4	0.6	4.0	0.036	0.2677	7.3333	0.0293	0.20	8.0	0.024
0.5	0.6	4.0	0.036	0.2527	7.4734	0.0283	0.20	8.0	0.024
0.6	0.6	4.0	0.036	0.2434	7.5660	0.0276	0.20	8.0	0.024
0.7	0.6	4.0	0.036	0.2369	7.6314	0.0271	0.20	8.0	0.024
0.8	0.6	4.0	0.036	0.2320	7.6798	0.0267	0.20	8.0	0.024
0.9	0.6	4.0	0.036	0.2283	7.7171	0.0264	0.20	8.0	0.024
1.0	0.6	4.0	0.036	0.2253	7.7467	0.0260	0.20	8.0	0.024
10.0	0.6	4.0	0.036	0.2024	7.9759	0.0242	0.20	8.0	0.024
100.0	0.6	4.0	0.036	0.2002	7.9976	0.0240	0.20	8.0	0.024

* Static private fishery and open-access fishery steady-state equilibrium values do not vary with the interest rate. These equilibrium values have been copied into every row for purposes of comparison.
Source of data: The Excel file *Comparative statics.xls*, to be found in the Additional Materials for Chapter 17

At the moment we just note these results. We shall explain them later.

17.8.4 The consequence of a dependence of harvest costs on the stock size

The specific functions used in this chapter for harvest quantity and harvest costs are $H = eES$ and $C = wE$ respectively. Substituting the former into the latter and rearranging yields $C = wE/eS$. It is evident from this that, under our assumptions, harvest costs depend on both effort and stock. However, suppose that the harvest equation were of the simpler form $H = eE$. In that case we obtain $C = wE/e$ and so costs are independent of stock. But in such rather unlikely cases, when harvesting costs do *not* depend on the stock size, $\partial C/\partial S = 0$ and so the steady-state PV-maximising condition (17.32) simplifies to $i = \mathrm{d}G/\mathrm{d}S$.

This is an interesting result. It tells us that a private PV-maximising steady-state equilibrium, where access can be controlled and costs do not depend upon the stock size, will be one in which the resource stock is maintained at a level where the rate of biological growth ($\mathrm{d}G/\mathrm{d}S$) equals the market rate of return on investment (i) – exactly what standard capital theory suggests should happen. This is illustrated in Figure 17.7. At the present-value-maximising resource stock size (which we denote by S_{PV}),

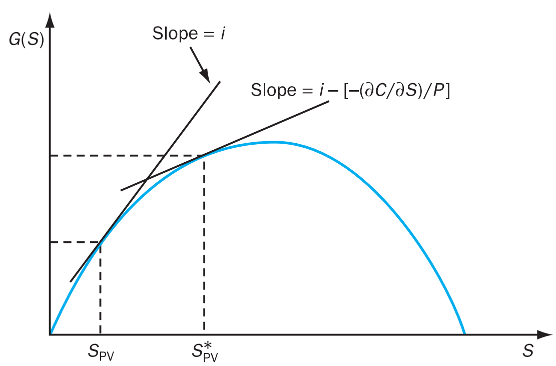


Figure 17.7 Present-value-maximising fish stocks with and without dependence of costs on stock size, and for zero and positive interest rates

$i = \mathrm{d}G/\mathrm{d}S$. It is clear from this diagram that as the interest rate rises, the profit-maximising stock size will fall. Some other results can also be obtained analytically (all of which are also evident by inspection of Figure 17.7).

- If $i = 0$ *and* $\partial C/\partial S = 0$ (that is, decision makers use a zero discount rate and harvesting costs do not depend on the stock size) then the stock rate of growth ($\mathrm{d}G/\mathrm{d}S$) should be zero, and so the present-value-maximising steady-state stock level would be the one generating the maximum sustainable yield, S_{MSY}. (In Figure 17.7, the straight line labelled 'Slope = i' would be horizontal and so tangent to the growth function at S_{MSY}.) Intuitively, it makes sense to pick the stock level that gives the highest yield in perpetuity if costs are unaffected by stock size and the discount rate is zero. However, this result is of little practical relevance in commercial fishing, as it is not conceivable that owners would select a zero discount rate.

- If $i > 0$ *and* $\partial C/\partial S = 0$ (that is, decision makers use a positive discount rate and harvesting costs do not depend on the stock size) then S_{PV} is less than the maximum sustainable yield stock, S_{MSY}. This follows from the fact that the equality $i = \mathrm{d}G/\mathrm{d}S$ can only be satisfied for positive i where $\mathrm{d}G/\mathrm{d}S$ is positive, which requires a stock size below S_{MSY}. Intuitively, with positive discounting and no stock effect on costs, the stock is drawn down below the maximum sustainable yield as future losses in income from higher harvests are discounted and there is no penalty from harvest cost increases. However, for many marine fisheries, it is likely to be the case in practice that harvesting costs will rise as the stock size falls, and so this result is unlikely to be of wide applicability.
- When total costs of harvesting, C, depend negatively on the stock size, the present-value-maximising stock is higher than it would be otherwise. This follows from the result that the discount rate i must be equated with $\mathrm{d}G/\mathrm{d}S - \{\partial C/\partial S\}/p)$, rather than with $\mathrm{d}G/\mathrm{d}S$ alone. Given that $\{\partial C/\partial S\}/p$ is a negative quantity, this implies that $\mathrm{d}G/\mathrm{d}S$ must be lower for any given interest rate, and so the equilibrium stock size must be greater than where there is no stock effect on costs. The intuition behind this is that benefits can be gained by allowing the stock size to rise (which causes harvesting costs to fall). This is also illustrated in Figure 17.7, which compares the present-value optimal stock levels with (S^*_{PV}) and without (S_{PV}) the dependence of costs on the stock size. This result is consistent with our findings in Table 17.6.
- In the general case where a positive discount rate is used and $\partial C/\partial S$ is negative, the steady-state present-value-maximising stock level may be less than, equal to, or greater than the maximum sustainable yield stock. Which one applies depends on the relative magnitudes of i and $(\partial C/\partial S)p$. Indeed inspection of Figure 17.7 shows the following:

$$i = -(\partial C/\partial S)p \Leftrightarrow S^*_{PV} = S_{MSY}$$

$$i > -(\partial C/\partial S)p \Leftrightarrow S^*_{PV} < S_{MSY}$$

$$i < -(\partial C/\partial S)p \Leftrightarrow S^*_{PV} > S_{MSY}$$

This confirms an observation made earlier in this chapter: a maximum sustainable yield (or a stock level S_{MSY}) is not in general economically efficient, and will only be so in special circumstances. For our illustrative numerical example, using baseline parameter values, it turns out that this special case is that in which $i = 0.05$; only at that interest rate does the PV-maximising model yield the MSY outcome.[25]

17.9 Dynamics in the PV-maximising fishery

Our analysis of the open-access fishery showed complex patterns of dynamic behaviour. Out of equilibrium, the processes of adjustment over time were (under baseline assumptions) characterised by oscillatory paths for stock, effort and harvest, with levels of these variables repeatedly over-shooting and under-shooting their steady-state values. In the case investigated with simple logistic growth, the dynamic adjustment processes were dynamically stable, eventually converging to equilibrium levels. For other sets of parameter values, other patterns of dynamic behaviour can be found, including fishery collapse and chaotic behaviour. (See Clark, 1990, and Conrad and Clark, 1987 for further details.)

These various dynamic processes reflect the unplanned and uncoordinated behaviour of fishing effort in an open-access fishery. Given the assumptions that have been made in this chapter about the

[25] You can verify this yourself using the spreadsheet *Comparative Statics.xls*. Note also that if P were chosen differently, then another value of i would be required to bring about an MSY solution.

PV-maximising fishery (and about the static private-property fishery) we would not expect to find those dynamic processes there. Even where there is a large number of profit maximising owners acting as if they were in a competitive market, the individual private property assumption is sufficient for owners to somehow or other coordinate their behaviour to be collectively rational. This rules out the intertemporal stock externality effect (whereby each boat's harvest reduces the future stock available for others) that exists in an open-access fishery.

A brief description of adjustment dynamics in PV-maximising private-property fisheries is given in Appendix 17.3. We suggest there that the efficient (wealth-maximising) paths to steady-state equilibrium outcomes will typically not be oscillatory. In fact, for the model examined in this section where the market price of fish is exogenously fixed, adjustment takes a particularly simple form, known as the most rapid approach path (MRAP). This is characterised by a simple rule

$$H_t = \begin{cases} 0 & \text{when } S_t < S^*_{PV} \\ H_{\text{MAX}} & \text{when } S_t > S^*_{PV} \end{cases}$$

That is, do no harvesting when stocks are less than the equilibrium value, and harvest at the maximum possible rate when stocks exceed the PV equilibrium.[26] Spence and Starrett, 1975, identify the conditions on the Hamiltonian which make MRAP optimal. Matters are more complicated where the price is endogenous, depending on the harvest rate. Relevant results, obtained using phase-plane analysis, are given in Appendix 17.3.

17.10 Encompassing the open-access, static private-property and PV-maximising fishery models in a single framework

For pedagogical reasons, this chapter has employed a variety of methods and graphical techniques to obtain results for the three fishery models that have been investigated. However, all of these can be related to one another and so what might appear to be three fundamentally different sets of models and results can all be reconciled with one another and encompassed within a single framework.

Consider first the two private-property fishery models. Earlier remarks have suggested that the static private-property fishery is a special case of the PV fishery – the case in which the interest rate is zero. Put another way, the steady-state PV-maximising fishery results collapse to those of the simple private fishery model when the interest (discount) rate is zero. This can be verified by substitution of the value $i = 0$ into the expressions for PV-maximising stock, effort and harvest levels (shown in Appendix 17.3). It will be found that they collapse to the simpler expressions for static private-property equilibria in Table 17.4. To understand intuitively why this happens, return to the dynamic asset-equilibrium condition given by equation 17.32:

$$ip = p\frac{\mathrm{d}G}{\mathrm{d}S} - \frac{\partial C}{\partial S}$$

When $i = 0$, this collapses to

$$\frac{\mathrm{d}G}{\mathrm{d}S}p = \frac{\partial C}{\partial S}$$

The left-hand side of this expression is the marginal revenue (with respect to stock changes) and the right-hand side is the marginal cost (with respect to stock changes). Profit-maximising equilibrium (in the static private-property fishery model without discounting) requires that these be equal. This is, of course, the standard result for any static profit-maximising model. Indeed, if the reader obtains the derivatives $\mathrm{d}G/\mathrm{d}S$ and $\partial C/\partial S$ for our particular model equations, substitutes these into the expression above, and then solves for the equilibrium level of stock, they will find that the result is identical to that derived in Section 17.4 (and listed in the column labelled 'Static private property' in Table 17.4).

[26] Discrete-time simulations, shown in the Excel workbook that accompanies this chapter, do not exhibit this MRAP form of adjustment for a private (nor for a PV-maximising) fishery. Instead, the workbook uses a discrete period-by-period adjustment rule in which effort increases (decreases) when marginal profit of effort is positive (negative). This rule generates a correct steady-state outcome, but the adjustment to it is slower than optimal, and does show some oscillatory behaviour. This corroborates an earlier comment that even for equivalent parameterisations discrete-time and continuous-time models can have qualitatively very different dynamic properties.

We recommend that you try this exercise; it can be checked against our solution, given in Appendix 17.2.

The open-access fishery model can also be brought into this encompassing framework. We can do so by observing that an open-access fishery can be thought of as one in which the absence of enforceable property rights means that fishing boat owners have an infinitely high discount rate. The interest rate of boat owners in an open-access fishery is, in effect, infinity, irrespective of what level the prevailing interest rate in the rest of the economy happens to be. We have already demonstrated that as increasingly large values of i are plugged into the PV-maximising solution expression, outcomes converge to those derived from the open-access model.

17.11 Socially efficient resource harvesting

As with all resource allocation decisions, there can be no guarantee that privately maximising decisions will be socially efficient (let alone socially optimal). In this section, we review some of the reasons why divergences may take place. First of all, though, we establish the conditions that must be satisfied for socially efficient harvesting.

Let r be the *social* consumption discount rate, and let BS denote social benefits from landed fish and CS denote social costs associated with landing fish. Socially efficient harvesting emerges as the solution to the maximisation problem

$$Max \int_0^\infty \{BS(H_t) - CS(S_t, H_t)\}e^{-rt}\mathrm{d}t$$

subject to

$$\frac{\mathrm{d}S}{\mathrm{d}t} = G(S_t) - H_t$$

and initial stock level $S(0) = S_0$. In this expression BS(H) denotes the social benefits as a function of quantity of fish landed, and CS(S, H) denotes the social costs as a function of stock size and harvest rate. The current-value Hamiltonian, L, for this problem is

$$L(H, S)_t = \mathrm{BS}(H_t) - \mathrm{CS}(S_t, H_t) + p_t(G(S_t) - H_t)$$

where p_t is the (social) shadow net price of a unit of the stock of the renewable resource. The necessary conditions for a maximum include

$$\text{(i)}\quad \frac{\partial L_t}{\partial H_t} = 0 = \frac{\mathrm{d}BS}{\mathrm{d}H_t} - \frac{\partial CS}{\partial H_t} - p_t \qquad (17.34)$$

$$\text{(ii)}\quad \frac{\mathrm{d}p_t}{\mathrm{d}t} = rp_t - p_t\frac{\mathrm{d}G}{\mathrm{d}S_t} + \frac{\partial CS}{\partial S_t} \qquad (17.35)$$

and the resource net growth equation

$$\text{(iii)}\quad \frac{\mathrm{d}S}{\mathrm{d}t} = G(S_t) - H_t$$

We continue to assume that the market price of fish is exogenously fixed at P.[27] Now consider also the following additional conditions:

(i) The (gross) market price, P, correctly reflects all social benefits (which then, together with the assumption of exogenously given prices, implies dBS/dH = P).
(ii) There are no fishing externalities on the cost side so that CS(S, H) = C(S, H).
(iii) The private and social consumption discount rates are identical, $i = r$.

If these additional conditions are satisfied, and are imposed on equations 17.34 and 17.35, the first-order conditions for social efficiency are identical to those for the PV-maximising fishery (equations 17.29 and 17.30). Hence the PV-maximising fishery is socially efficient under the set of conditions we have just described. But, of course, it also follows that if one or more of those conditions is *not* satisfied, private fishing will not be socially efficient. We investigate next some reasons why such a divergence might arise.

17.11.1 Externalities in the benefits function

The first case is that in which social benefits depend not only on the size of the resource harvest but

[27] Be careful in these expressions to distinguish between the use of upper-case P (which denotes the gross price of a unit of landed fish) and lower-case p (which denotes the shadow price, or net price, of a unit of landed fish). In equations 17.34 and 17.35 it is lower-case p rather than upper-case P which appears.

also on the level of the resource stock. For many biological species that are harvested, intentionally or unintentionally (as by-catch), it is evident that society does place a value on the existence of these species and is concerned about the number of them that do exist. This is clearly true for many large animals such as big cats, whales and apes, and surely extends much more widely than that. In this case, the social-welfare-maximising problem should be generalised to

$$\text{Max} \int_0^{\infty} \{\text{BS}(H_t, S_t) - \text{CS}(H_t, S_t)e^{-rt}\text{d}t$$

subject to

$$\frac{\text{d}S}{\text{d}t} = G(S_t) - H_t$$

Note that S now enters the social benefits function (in which it had not appeared previously). Intuition suggests that the optimal solution to this model will be different from that obtained from the model where benefits only depend on the harvest rate, H. This intuition is correct. Assuming that utility is an increasing function of S, the solution will in general be one in which the optimal stock level is higher, reflecting the positive utility that the resource stock generates. We invite the reader to work through the maths to obtain this result. To check your analysis, we have placed such a derivation in a separate document on the Companion Website, with the title *Stock Dependent Utility.doc*.

More generally, society is likely to have multiple objectives which are not well represented by the private harvester's own objective function (which will tend to be dominated by catch quantity considerations). This is very important for many terrestrial resources, particularly woodlands and forests as we shall see in the next chapter. But it also applies to marine resources. For example, society may have an interest in the maintenance of population diversity or genetic diversity; it may be willing to pay a larger risk premium to ensure high resistance to disease among marine organisms; or it may prefer to maintain stock levels much higher than would private harvesters – at a safe minimum standard – in response to uncertainty and the threats of catastrophic change. All these could be thought of as additional arguments that would appear in the social objective function (but which would not usually enter private profit functions).

Alternatively, we might choose to model some or all of the externalities in terms of costs. That is, private owners are likely to fail to make adequate provision for the full social opportunity costs of their activities, and so externalities are not being internalised. This leads us to the next category of sources of inefficiency.

17.11.2 Externalities in the fishery production function

A second source of social inefficiency arises from externalities operating through the fishery production function. There are two important types of harvesting externality. First, it often happens that resource harvesting inadvertently destroys other species. One example of this is that of dolphins getting caught in tuna nets. Chapter 10 elaborates on that problem. A second example is beam trawling, in which a net is weighed down to the sea bed by heavy beams and is trawled along the sea bed to catch bottom-feeding fish such as cod. This causes immense damage to other sea-bed creatures, and can cause those populations to collapse. Each of these is a classic externality problem and so outcomes are most unlikely to be efficient in such cases. Some form of regulation of fishing practices seems to be appropriate here. We return to this matter later.

A second kind of externality is often known as a 'crowding' diseconomy. Suppose that each boat's harvest depends on its effort and on the effort of others. Then each boat's catch imposes a contemporaneous external cost on every other boat. In effect, boats are getting in each other's way. When any boat is fishing, the costs of harvesting a given quantity of fish become higher for all other boats. This externality drives the average costs of fishing for the fleet as a whole above the marginal costs of an individual fisher. Put another way, if a crowding effect of this kind exists, the function $C(H, S)$ from the point of view of an individual boat operator may differ from the function $\text{CS}(H, S)$ from the social point of view.

Crowding externalities could occur under any form of fishing regime, but the likelihood of their occurrence will depend on what institutional conditions apply. Crowding is most likely under conditions of open access where there is an absence of any coordination of behaviour between individual fishing firms. In the case of a private property (or PV-maximising) fishery one would expect that some mechanism exists by means of which effort is coordinated in the common interest. In these institutional circumstances, it would be sensible to assume that the size of the fishing fleet as a whole (and the spatial and temporal patterns of fishing) would be optimally chosen. The optimal size of fleet would balance the additional benefits of extra boats against the additional external costs of extra boats. The crowding diseconomies become internalised in this way, leading to efficient outcomes. Alternatively, if the fishery was in private-property ownership but failed to coordinate activity of individual fishing firms, then if it were carefully and effectively *regulated* it is conceivable that such regulation might also internalise the externality.[28]

Under conditions of open access, there is virtually no possibility that crowding effects would be internalised by the actions of fishermen alone. The kind of coordination we referred to above cannot happen in the competitive struggle to grab fish. It is important to note that the crowding diseconomy we have just referred to is entirely different from the 'stock externality' effect which arises in open access. Crowding externalities are contemporaneous; stock externalities are intertemporal. The latter exist when the taking of fish today imposes additional costs in the future by virtue of the reduced future stock size. We have already remarked that this kind of externality fundamentally distinguishes the open-access and private-property (and PV-maximising) fishery cases. Modellers typically assume that the stock externality will not be internalised in open-access conditions but will be in a private fishery.

Taken as a whole, one is led to a strong presumption that open-access fisheries are both privately and socially inefficient. Social inefficiency arises from the likely presence of crowding externalities and from the near-inevitability of intertemporal stock externalities. From a private perspective, the open-access fishery is inefficient because of its rent-dissipating behaviour, which is in turn a reflection of the externalities that have just been referred to. Indeed, there is ample evidence that fishing effort is massively excessive and inefficient in many open-access fisheries throughout the world. Almost certainly, unregulated open access fisheries would consist of more harvesters and more harvesting capital than is economically efficient. However, it is still possible that some form of regulation might achieve the required coordination. We discuss this further below.

17.11.3 Difference between private and social discount rates

Maximised social and private net benefits also diverge when social and private discount rates differ. We have already remarked that this is one way of explaining the excessive harvesting that takes place in open-access fisheries, but it should be clear that the problem is more wide-reaching than that case alone.

It is also worth noting here that the steady-state stock size decreases with an increasing discount rate. There are reasons why even sole or private-property regime owners might have high discount rates. So to the extent that private discount rates exceed an appropriate social rate, steady-state resources stocks will be inefficiently low. Low steady-state stock levels may also, of course, be undesirable on grounds other than economic efficiency criteria, most notably in terms of adverse impacts on resource sustainability.

17.11.4 Monopolistic fisheries

The existence of monopoly ownership of a fishery may also generate inefficient outcomes. A resource market is monopolistic if there is one single price-making harvester. It is known from standard

[28] It is of interest to note that the Schaefer (1957) form of fishery production function we have used in this chapter ($H = eES$) rules out the existence of crowding externalities. For a given stock size, H will change in proportion to changes in E. In other words, the marginal product of effort is constant, precluding crowding effects. So to be able to model formally crowding externalities, a more general form of fishery function than that employed in this chapter would be required.

micro-economic theory that marginal revenue exceeds marginal cost in a monopolistic market equilibrium.[29] A monopoly owner would tend to harvest less each period, and sell the resource at a higher market price, than is socially efficient. Therefore, if a renewable resource were harvested under monopolistic rather than competitive conditions, an economically inefficient harvesting level may result. The qualifier 'may' in the previous sentence arises from second-best considerations. In a world where there are other market failures pushing harvest rates to excessive levels, monopoly harvests may be closer to the second-best efficient allocation than those from 'competitive' PV-maximising fisheries. So, as with non-renewables, the monopolist could be the conservationist's friend, albeit at the price of some inefficiency.

17.12 A safe minimum standard of conservation

Our discussion of the 'best' level of renewable-resource harvesting has focused almost exclusively on the criterion of economic efficiency. However, if harvesting rates pose threats to the sustainability of some renewable resource (such as North Atlantic fisheries or tropical primary forests) or jeopardise an environmental system itself (such as a wildlife reserve containing extensive biodiversity) then the criterion of efficiency may be inappropriate. Even in a deterministic world in which population growth rates are known with certainty the pursuit of an efficiency criterion is not sufficient to guarantee the survival of a renewable resource stock or an environmental system in perpetuity, particularly when resource prices are high, harvesting costs are low, and/or discount rates are high. Where biological systems are stochastic, or where uncertainty is pervasive, threats to sustainability are even more pronounced.

Many writers – some economists, but particularly non-economists – argue that correcting market failure and eliminating efficiency losses should be given secondary importance to the pursuit of sustainability. This would suggest that policy be targeted to the prevention of species extinction or the loss of biological diversity whenever that is reasonably practical. Efficiency objectives can be pursued within this general constraint.

Such considerations brings us back to the principle that policy be oriented around the criterion of a safe minimum standard of conservation (SMS), an idea examined earlier in Chapter 13. How does this idea apply to renewable resource policy? A strict version of SMS would involve imposing constraints on resource harvesting and use so that all risks to the survival of a renewable resource are eliminated. This is unlikely to be of much practical relevance. Virtually all human behaviour entails some risks to species survival, and so a strict SMS would prohibit virtually all economic activity. In order to make the concept usable, it is necessary to impose weaker constraints, so that the adoption of an SMS approach will entail that, under *reasonable* allowances for uncertainty, threats to survival of valuable resource systems are eliminated, provided that this does not entail excessive cost. For decisions to be made that are consistent with that weaker criterion, judgements will be necessary particularly about what constitutes 'reasonable uncertainty' and 'excessive cost', and which resources are deemed 'sufficiently valuable' for application of the SMS criterion.

Let us explore the concept of an SMS in this context by following the exposition in a paper by Randall and Farmer (1995). Suppose there is some renewable resource the expected growth of which over time is illustrated by the curve labelled 'Regeneration function' in Figure 17.8. The function shows the resource stock level that will be available in period $t + 1$ (S_{t+1}) for any level of stock that is conserved in period t (S_t). Notice that the greater is the level of stock conservation in period t, the higher will be the stock level available in the following period.[30]

Randall and Farmer restrict their attention to sustainable resource use, interpreting sustainability to mean a sequence of states in which the resource

[29] Similar conclusions also apply to imperfectly competitive markets in which a small number of harvesters dominate the industry.

[30] The non-linear relationship shown here is plausible for many types of biological resource, but will not be a good representation for all such resources.

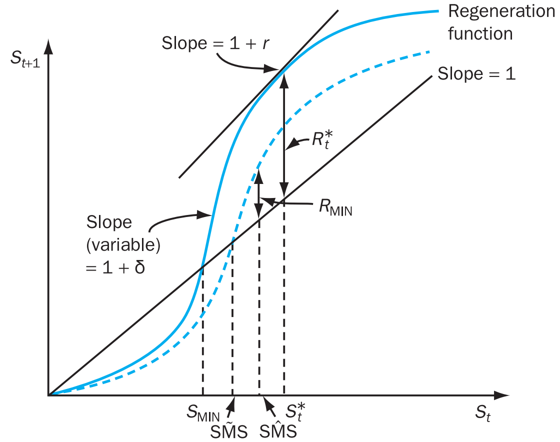


Figure 17.8 A safe minimum standard of conservation
Source: Adapted from Randall and Farmer (1995)

stock does not decline over time. Therefore, only those levels of stock in period t corresponding to segments of the regeneration function that lie on or above the 45° line (labelled 'slope = 1') constitute sustainable stocks. The minimum *sustainable* level of stock conservation is labelled S_{MIN}.

The efficient level of stock conservation is S_t^*. To see this, construct a tangent to the regeneration function with a slope of $1 + r$, where r is a (consumption) social discount rate, and let δ denote the rate of growth of the renewable resource.[31] At any point on the regeneration function, a tangent to the function will have a slope of $1 + \delta$. For the particular stock level S_t^* we have

$$1 + r = 1 + \delta \quad \text{or} \quad r = \delta$$

and so the rate of growth of the renewable resource, δ, is equal to the social discount rate, r. This is a condition for *economically efficient* steady-state renewable resource harvesting that we have met earlier in Section 17.7.1. That is, it is the form that equation 17.32 takes in steady-state equilibrium where harvest costs are independent of stock size. Note also that at the efficient stock level, the amount of harvest that can be taken in perpetuity is R_t^*. By harvesting at this rate, the post-harvest stock in period $t + 1$ is equal to that in period t, thus satisfying the sustainability requirement.

Now suppose that the regeneration function is subject to random variation. For simplicity, assume that the worst possible outcome is indicated by the dotted regeneration function in Figure 17.8. At any current stock, the worst that can happen is that the available future stock falls short of the expected quantity by an amount equal to the vertical distance between the solid and dotted functions. Now even in the worst outcome, if $\tilde{SMS}$ is conserved in each period, the condition for perpetual sustainability of the stock will be maintained. We might regard $\tilde{SMS}$ as a stock level that incorporates the safe minimum standard of conservation, reflecting the uncertainty due to random variability in the regeneration function. Put another way, whereas S_{MIN} is an appropriate minimum stock in the absence of uncertainty, $\tilde{SMS}$ takes account of uncertainty in a particular way.

In fact, $\tilde{SMS}$ is not what Randall and Farmer propose as a safe minimum standard of conservation. They argue that any sustainable path over time must involve some positive, non-declining level of resource harvesting and consumption in every period. Suppose that R_{MIN} is judged to be that minimum required level of resource consumption. Then Randall and Farmer's safe minimum standard of conservation is $\hat{SMS}$. If the stock in period t is kept from falling below this level then, even in the worst possible case, R_{MIN} can be harvested without interference with sustainability.

The SMS principle implies maintaining a renewable resource stock at or above some safe minimum level such as $\tilde{SMS}$. In Figure 17.8, there is no conflict between the conservation and efficiency criteria. The safe minimum standard of conservation actually implies a lower target for the resource stock than that implied by economic efficiency. This will not always be true, however, and one can easily imagine circumstances where an SMS criterion implies more cautious behaviour than the economically efficient outcome.

Finally, what can be said about the qualification that the SMS should be pursued only where it does

[31] Strictly speaking, δ is the derivative of the amount of resource growth with respect to the stock size; it is what we have earlier written as dG/dS.

not entail excessive cost? Not surprisingly, it is difficult to make much headway here, as it is not clear how one might decide what constitutes excessive cost. Randall and Farmer suggest that no society can reasonably be expected to decimate itself. Therefore, if the SMS conflicted with the survival of human society, that would certainly entail excessive cost. But most people are likely to regard costs far less than this – such as extreme deprivation – as being excessive. Ultimately, the political process must generate views as to what constitutes excessive costs.

17.13 Resource harvesting, population collapses and the extinction of species

The absence of enforceable private property rights creates an *a priori* presumption that outcomes will be economically inefficient. But it does not *necessarily* imply that renewable resources will fail to be harvested in an efficient and sustainable manner. Nor does it imply anything about the inevitability that a resource will be harvested to the point of complete population collapse or species extinction.

It is important to distinguish between 'common property' and 'open access' resources. Where resources are subject to common property, property rights are not vested in individuals but rather in communities. Here, relationships of trust and social norms may be sufficiently well developed and entrenched to create patterns of resource exploitation that are both sustainable and rational for the group as a whole (see for example Bromley, 1999 and Ostrom, 2000a, 2000b). In contrast, open access is usually taken to mean the absence of any such binding norms with a tendency for exploitation to take place under conditions of 'free-for-all' individualistic competition.

It is possible that any resource stock could be harvested to exhaustion, or a species driven to extinction, under open access. Indeed, this is true under almost any regime, including those with enforceable private-property rights. Nevertheless, it remains true that open-access conditions increase the probabilities of those outcomes occurring. Why is this? The main reason is that in these circumstances there is no means by which collectively rational management of harvesting can take place. Even where what should be done is evident, an institutional mechanism to bring this about is missing. When one compares open-access outcomes with those of private-property regimes, analysis suggests that harvest rates are typically higher under conditions of open access; other things being equal, the greater are harvesting rates, the higher is the likelihood of extinction.

Another way of thinking about this is in terms of stock externality effects and economic inefficiency. Open-access harvesting programmes are inefficient because resource harvesters are unable to appropriate the benefits of investment in the resource. If a single fisher were to defer the harvesting of some fish until a later period, all fishers would benefit from this deferral. It would be in the interests of all if a bargain were made to reduce fishing effort by the industry. However, the conditions under which such a bargain could be made and not reneged upon are very unlikely to exist. Each potential bargainer has an incentive to free-ride once a bargain has been struck, by increasing his or her harvest while others reduce theirs. Something akin to a Prisoner's Dilemma is operating. Moreover, even if all existing parties were to agree among themselves, the open-access conditions imply that others could enter the market as soon as rents became positive. Open-access resources thus have one of the properties of a public good – non-excludability – and this alone is sufficient to make it likely that markets will fail to reach efficient outcomes.

The recent history of the Peruvian anchovy fishery, described in Box 17.4, illustrates such an outcome. Another facet of open access is also shown in Box 17.4. Overfishing does not usually happen because of ignorance; it tends to result from the forces of competition in conditions where access is poorly regulated. The New England fisheries demonstrate that self-regulation on the part of fishermen may do little to overcome the consequences of open access.

It will be useful to draw together some of the results obtained in this chapter that relate to the possibilities of major population declines or the extinction of species. Human activity can have adverse effects on biological resources in a variety of ways.

Box 17.4 A story of two fish populations

One species of fish – the Peruvian anchovy – and one group of commercial fish – New England groundfish – provide us with case studies of the mismanagement and economic inefficiency which often characterise the world's commercial fisheries. In this box, we summarise reviews of the recent historical experiences of these two fisheries; the reviews are to be found in WR (1994), chapter 10.

Peruvian anchovy are to be found in the Humboldt Upswelling off the west coast of South America. Upswellings of cold, nutrient-rich water create conditions for rich commercial fish catches. During the 1960s and 1970s, this fishery provided nearly 20% of the world's fish landings. Until 1950, Peruvian anchovy were harvested on a small scale, predominantly for local human consumption, but in the following two decades the fishery increased in scale dramatically as the market for fishmeal grew. The maximum sustainable yield (MSY) was estimated as 9.5 million tonnes per year, but that figure was being exceeded by 1970, with harvests beyond 12 million tonnes. In 1972, the catch plummeted. This fall was partially accounted for by a cyclical natural phenomenon, the arrival of the El Niño current. However, it is now recognised that the primary cause of the fishery collapse (with the catch down to just over 1 million tonnes in the 1980s) was the conjunction of overharvesting with the natural change associated with El Niño. Harvesting at rates above the MSY can lead to dramatic stock collapses that can persist for decades, and may be irreversible (although, in this case, anchovy populations do now show signs of recovery).

The seas off the New England coast have been among the most productive, the most intensively studied and the most heavily overfished in the world since 1960. The most important species in commercial terms have been floor-living species including Atlantic cod, haddock, redfish, hake, pollock and flounder. Populations of each are now near record low levels. Although overfishing is not the only contributory factor, it has almost certainly been the principal cause of stock collapses. The New England fisheries are not unusual in this; what is most interesting about this case is the way in which regulatory schemes have failed to achieve their stated goals. In effect, self-regulation has been practised in these fisheries and, not surprisingly perhaps, regulations have turned out to avoid burdening current harvesters. This is a classic example of what is sometimes called 'institutional capture': institutions which were intended to regulate the behaviour of firms within an industry, to conform with some yardstick of 'the common good', have in effect been taken over by those who were intended to be regulated, who then design administrative arrangements in their own interest. The regulations have, in the final analysis, been abysmal failures when measured against the criterion of reducing the effective quantity of fishing effort applied to the New England ground fisheries.

Long-term solutions to overfishing will require strict quantity controls over fishing effort, either by direct controls over the effort or techniques of individual boats, or through systems of transferable, marketable quotas. We investigated some of these instruments in Chapter 6 and do so further later in this chapter.

Two general classes of effects can be distinguished. One operates on particular species (or local populations of a species) that are the direct targets of harvesting activity. In this category we shall also include effects on related species or populations that are strongly dependent on or interrelated with the targets. The second class concerns more widely diffused, indirect impacts on systems of biological resources, induced in the main by land-use changes that lead to disruption of ecosystems. Much of this generalised impact on biological resources is brought together under the rubric of decline of biological diversity. We consider the causes of biodiversity decline in the document '*What is causing the loss of biological diversity*' available from the Companion Website.

Property rights have important consequences for patterns of resource exploitation. In general, harvesting effort will be higher and stock sizes lower where a renewable resource is open-access than when enforceable property rights exist. But stronger claims are often made. In particular, it is sometimes argued that populations of renewable resources will inevitably be degraded, suffer serious collapse, or even be driven to zero under open-access conditions. The *inevitability* of these events finds no support in the models we have examined so far. Indeed, in examining the steady-state equilibrium of an open-access fishery, the model examined earlier in this chapter showed that some configurations of parameters are consistent with the fishery having a maximum-sustainable-yield stock size. And for some parameter values (particularly those relating to harvesting costs or net price of the resource) steady-state stock could be higher, possibly as large as S_{MAX}, the environmental carrying capacity of the resource.

To confirm this, look again at Figures 17.3 and 17.6. Consider, for example, the consequence of an increase in w. The ray which plots $H = (w/P)E$ in Figure 17.6 shifts anticlockwise, and so leads to lower effort. Lower effort rotates the *eES* curve in Figure 17.3 clockwise, leading to a higher steady-state stock. For a sufficiently high value of w, no positive steady-state fishing effort will be profitable, and the population will rise to its environmental carrying capacity. Clearly, there is nothing inevitable about population collapses in open-access conditions. Much depends on economic factors that may be favourable or unfavourable to large population sizes. Nevertheless, the possibility that a population may be driven to zero is greater

- when the resource is harvested under conditions of open access than where enforceable property rights prevail;
- the higher is the market price of the harvested resource;
- the lower is the cost of harvesting a given quantity of the resource;
- when prices are endogenous, the more that market price rises as catch costs rise or as harvest quantities fall;
- the lower the natural growth rate of the stock;
- the lower the extent to which marginal extraction costs rise as the stock size diminishes;
- the higher is the discount rate.

Even under private-property conditions an optimal harvesting programme may drive a fish stock to zero. This is most likely where the prey is simple to catch even when the stock approaches a critical minimum threshold level, and where the harvested resource is very valuable. In this case, the optimal harvest level could exceed biological growth rates at all levels of stock. However, the literature that examines dynamic models of commercial resource harvesting suggest that species extinction, while being possible in principle, is likely only under very special circumstances. For example, Clark (1990) shows that a privately optimal harvesting programme (where access to the resource is controlled) may be one in which it is 'optimal' to harvest a resource to extinction, but also demonstrates that this is highly improbable.

As the analyses in this chapter have in the main been derived from a simple logistic biological growth model one should be wary about claiming too much generality for our results. Clearly, other biological growth functions could have generated different results. Four matters are noteworthy here:

- The biological growth function may have a positive (and possibly quite large) minimum sustainable population size, as in Figures 17.1(b) and (c). If for any reason the stock falls below this level, the population cannot survive. Many large mammal species appear to be cases in point. Moreover, as we explained earlier, renewable resource models in which growth exhibits critical depensation tend to be characterised by unstable dynamics that can often drive populations below minimum sustainable stock levels. Extinction of species is more likely, other things being equal, where the critical minimum threshold population size is relatively large. In this case, a greater proportion of possible harvesting programmes will lead to harvest rates that cannot be maintained over time.
- The existence of uncertainty plays a very important role. Uncertainty may relate to the size of the minimum threshold population, to the actual stock size, or to the current and forecast harvesting rates. If errors are made in estimating any of these magnitudes, then it is clear that the likelihood of stock extinction is increased.
- The growth process may be stochastic (have a random component); stochastic shocks or disturbances might lead to population collapses. Our presentation has assumed that all functions

are deterministic – for given values of explanatory variables, there is a unique value of the explained variable. But many biological and economic processes are stochastic, with chance factors playing a role in shaping outcomes. In these circumstances, there will be a distribution of possible outcomes rather than one single outcome for given values of explanatory variables. We discussed some consequences of risk and uncertainty in Chapter 13.

- Populations are often interdependent in ecosystems. Changes in other populations may lead to the collapse of some population in which we are interested. The word file *Models of Biological Interaction* available from the Companion Website examines this matter at some length.

There are other reasons, too, why we should be cautious. Much discussion of renewable resources is framed largely in terms of steady-state or equilibrium outcomes. But systems are not always (or perhaps not even very often) in such a state. Indeed, simulations using discrete-time renewable resource models (see, for example, the Excel simulations described in *Fishery dynamics.xls* available on the Companion Website) demonstrate that an open-access fishery can be in disequilibrium over very long periods of time even where parameters and environmental conditions are unchanging. Where those factors are frequently changing – as we would expect to find in practice – disequilibrium states will prevail more or less indefinitely, and it is very difficult to predict outcomes. It is clearly possible that harvest rates may be persistently above natural population growth rates. Also, ignorance, uncertainty or institutional failure could lead to a population falling below its minimum threshold size even where that event is (*ex post*) irrational.

Moreover, the simulations show that the existence of a steady-state equilibrium is no guarantee that such a state may ever be attained. Again, this result is particularly relevant for discrete-time population dynamics; the Excel simulations described in *Fishery dynamics.xls* demonstrate that the population might collapse to its minimum threshold size (and never recover) even though a positive steady-state equilibrium stock level exists. Non-targeted species may be casualties in this process too. Many forms of resource harvesting, particularly marine fishing, directly or indirectly reduce stocks of other plants or animals that happen to be in the neighbourhood, or which have some biologically complementary relationship with the target resource.

Finally, we note that even a casual inspection of the available evidence suggests that much resource harvesting does not fit comfortably with the theoretical models we have outlined, nor does it appear to have the consequences that those models predict. The world is experiencing extensive losses of many renewable resource-population stocks, and unprecedented rates of species extinction. Some examples of these phenomena, looking particularly at cases where harvesting rates of target animal populations have been high relative to natural rates of regeneration, are presented in the file *Population Collapses* available from the Companion Website.

17.14 Renewable resources policy

National and international policy regarding the use and conservation of renewable resources has multiple objectives. In this mix, economic efficiency, biological and ecological sustainability, and regional employment and cultural support all appear to have substantial weight. Efficiency goals have been the main focus of this chapter. When the use of resources is economically inefficient, there are potential welfare benefits to be obtained from policy which promotes efficiency improvements.

This suggests that policy may be directed towards removing externalities, improving information, developing property rights, removing monopolist industrial structures, and using direct controls or fiscal incentives to alter rates of harvesting whenever there is reason to believe that harvesting programmes are inefficient. Efficiency gains, in the form of improved intertemporal resource extraction programmes, may also be obtained if government assists in the establishment of forward or futures markets. As we saw in Chapter 4, efficient outcomes are not possible in general unless all relevant markets exist. The absence of forward markets for most natural resources suggests that it is most unlikely that extraction and harvesting programmes will be intertemporally efficient. Chapters 6 and 7 examined the policy instruments available to attain environmental pollution targets, and much of what was written there is relevant for our present discussion. We will examine two particular instances of incentive-based policy instruments later in this section.

The provision of information matters not only to the pursuit of efficiency but also to biological and ecological sustainability. Given that uncertainty is so great in matters relating to natural resources, government's role in the provision of information is crucial. In the case of commercial fisheries, for example, individual fishermen will not be in a position to know, in quantitative terms, how previous and current behaviour has affected and is likely to affect the population levels of relevant species. The consequences of cyclical natural phenomenon such as the El Niño current will similarly be largely unpredictable by individual agents. Obtaining this kind of information requires a significant monitoring and research effort which is unlikely to be undertaken by the industry itself, unless its institutional structure facilitated group-rational enforceable property rights. Even if it were obtained privately, the dissemination of such information would probably be sub-optimal, as those who devote resources to collecting the information may well seek to limit its availability to others.

17.14.1 Command and control regulations

Most of the world's fisheries have for long been, and remain, open-access fisheries. Where jurisdictions have intervened in what were formerly open access fisheries, by far the most common form of regulation has been command and control. The large set of restrictions and regulations that fall under this heading can be loosely classified in the following way:

1. Regulations aimed at reducing fishing effort. Examples include restrictions on the boat size or other capital equipment used by fishermen, closed fishery seasons, limits on days of fishing permitted per boat.
2. Restrictions on fishing gear and mesh or net size, aimed at controlling the qualitatitive nature of the catch. Particular targets have included protection of juvenile fish, reduction of by-catch and catch discards, and reducing environmental damage associated with harvesting.
3. Spatial restrictions on harvesting activity, aimed at reducing conflict among fishing operators.
4. Fleet size reductions.
5. Quantity restrictions on catches per boat and/or per total fishery (with some associated schema for allocation a total quota among potential participants).

This is the centrepiece of fishing regulation in the European Union, for example, in the Total Allowable Catch system. A difficulty with the first three of these command and control approaches is that objectives are reached at the cost of decreased harvesting efficiency (in effect, a reduction in the catch coefficient, e, in equation 17.7) and/or large financial costs to fishing operators. These controls deliberately impose economic inefficiency on the industry, in an effort to reduce harvest sizes, and so cannot be cost-efficient methods of attaining harvest reduction targets.

The fourth approach – fleet size reduction – may avoid cost-inefficiencies if the fishing industry capital stock is reduced to its optimal (cost-minimising) level. Governments have tried to attain this by incentives for firms to leave the industry. And to the extent that these incentives have been successful, this leads to capacity being cut in the fishery that the regulators have power over. But there may be externalities spilling over to other fisheries; capital reductions in regulated fisheries (particularly in those of the industrialised economies) result in older vessels being sold cheaply to fisheries elsewhere; the effective capacity in a more widely defined set of fisheries may not actually be cut. It has also been argued that fleet size reductions brought about by incentives on incumbents to leave may have little net effect on fishing effort in a state of generalised over-capacity when reduced effort by incumbent and regulated operators is just matched by increased effort from other fishery firms (either incumbents

Box 17.5 Quota restrictions in the Pacific halibut fishery

Gordon (1954) provides an interesting account of the use of quota restrictions in the Pacific halibut fishery. During the 1930s, Canada and the USA agreed to fixed catch limits. For many years, the scheme was hailed as an outstanding success; with catch per unit effort quantities rising over two decades, it was one of the few quota schemes to have achieved this goal.

However, Gordon shows that the improvements were not the result of quotas, but of a natural cyclical improvement in Pacific halibut stocks. Catches rose rather than fell during the period when quotas were introduced. Even then, the total catch taken was only a small fraction of the estimated population *reduction* prior to the introduction of regulation.

Furthermore, the efficiency loss of the regulations was enormous. The actual duration of the fishing season (the time span until quotas were met) fell from six months in 1933 to between one and two months in 1952. Despite their widespread use, these quantitative restrictions on either effort or catch have very little justification in either economic or biological terms.

who were not attracted by the available incentives to leave or firms in other fisheries looking for better resource opportunities to exploit). Be that as it may, one may counter by making the point that if there is still generalised over-capacity, then capacity hasn't been cut enough.

The fifth approach, quantity of catch restrictions, may be attractive if enforceable, but its record of achievement is extremely poor. Box 17.5 shows that even one of the few examples of quota schemes that was thought to be successful turned out not to be so on closer examination. The widespread failure of catch quota instruments is partly the consequence of the mobile and geographically dispersed nature of the industry and its prey. That is being overcome by modern electronic-based methods of monitoring, control and surveillance. However, in the final analysis, this instrument – like others in the list – is flawed by its focus on proximate causes. It fails to tackle a more fundamental underlying cause. This has its roots in limits to the effectiveness of the state. In many countries, the state has taken national property rights in marine renewable resources, but it has not used those rights properly. Often states have moved to convert genuine open access to territorial waters, but then find it politically difficult to do what might be needed, such as cut fleet sizes. Similar issues arise at the supra-national level where the difficulties of carrying out effective regulation are magnified by international coordination problems. The European Common Fisheries Policy may well be a case in point. An analysis of this is available on the Companion Website.

17.14.2 Incentive-based instruments: a landing tax

Compliance with regulations is often so poor, and illegal, unregulated and unreported (IUU) fishing so widespread, that command and control techniques may simply fail to reach their objectives at all. Not surprisingly, this has led to a search for alternative policy instruments. We have seen in several earlier parts of this book that a tax instrument can sometimes internalise an externality so as to bring about an efficient solution. The 'problems' associated with open-access fisheries can also be interpreted as externality problems. In this section we show how – at least in principle – an open-access fishing industry might be induced to harvest in a socially efficient manner through the use of a tax instrument.

Specifically, the tax being looked at here is a fish landings tax. It is levied at a fixed rate per unit of the fish resource landed. How should the level of such a tax be set? Consider the steady-state present-value-maximising condition 17.32:

$$ip = p\frac{\mathrm{d}G}{\mathrm{d}S} - \frac{\partial C}{\partial S} \tag{17.36}$$

Remember that p in this equation is the net price of a unit of the fish resource. Rewrite this equation in the form

$$p = \frac{\frac{\mathrm{d}G}{\mathrm{d}S}p}{i} - \frac{\frac{\partial C}{\partial S}}{i} \tag{17.37}$$

How can we interpret this equation? Consider a decision to *not* harvest an additional unit of the resource. The left-hand side of this expression is the net price of the fish; it is, therefore, the rent forgone by not harvesting that unit. As that rent would be obtained immediately if the fish were harvested, the left-hand side is already in present-value terms. The right-hand side of 17.37 consists of the present value of the benefits obtained by not harvesting that unit of fish. This benefit has two components:

- $[(\mathrm{d}G/\mathrm{d}S)p]/i$ is the present value of the extra biological growth that would result from leaving the fish unharvested (including any new growth of fish that could be attributed to it);
- $-(\partial C/\partial S)/i$ is the present value of the reduced fishing costs that would result from leaving the fish unharvested, noting that the costs per unit harvest will generally be smaller the larger is the fish stock level.

How do we know that these two terms are in present values? It follows from the expression for the present value of an infinite-duration annuity. The present value of an infinite annuity of A at an interest rate i is A/i. As we are considering steady states only here, the numerator term in each of the two components is an infinitely repeated benefit or cost. Dividing one repetition of either by the interest rate gives us its present value, therefore.

The right-hand side of equation 17.37 is often known as the *user cost* of fishing. This is taken into account in optimal (present-value-maximising) harvesting choices. However, in an open-access fishery, even though each fisherman may well be aware of these two components of costs, the user cost of fishing will *not* be taken into account in choices about effort or harvesting. Why not? The open-access property gives the fisherman no incentive to leave fish in the sea (to encourage more growth in the future). Others would simply harvest any fish left by one fisherman. The fisherman would be unable to appropriate the fruits of his investment in the future. Each fisherman in effect sees the present value of these future benefits as zero, even though collectively they would not be zero if all abstained together.

Recall that in open-access conditions, fishing effort expands until a point is reached at which the gross (or market) price minus *average* cost is zero. That is, in open access equilibrium $p = P - C/H = 0$ where p is the *net* price of the fish resource, P is its gross or market price, and C/H is the average cost per unit of fish caught. However, the condition that must be satisfied by optimal (present-value-maximising) fishing is, using equation 17.37:

$$p - \left(\frac{\frac{\mathrm{d}G}{\mathrm{d}S}p}{i} - \frac{\frac{\partial C}{\partial S}}{i}\right) = 0$$

By substitution for the net price p from the open-access equilibrium condition, we obtain

$$\left(P - \frac{C}{H}\right) - \left(\frac{\frac{\mathrm{d}G}{\mathrm{d}S}p}{i} - \frac{\frac{\partial C}{\partial S}}{i}\right) = 0 \tag{17.38}$$

where C/H is the average cost of each fish harvested. But now suppose that the open-access fisherman is required to pay a tax t^* on each unit of fish landed equal to the user cost of fishing *plus* the average cost of fishing (C/H) *minus* the marginal cost of fishing ($\partial C/\partial H$). That is

$$t^* = \left(\frac{\frac{\mathrm{d}G}{\mathrm{d}S}p}{i} - \frac{\frac{\partial C}{\partial S}}{i}\right) + \frac{C}{H} - \frac{\partial C}{\partial H} \tag{17.39}$$

Then, the post-tax open-access equilibrium condition would be amended so that the post-tax net price is zero. That is,

$$\left(P - \frac{C}{H} - t^*\right) = \left(P - \frac{C}{H}\right) - \left(\frac{\frac{\mathrm{d}G}{\mathrm{d}S}p}{i} - \frac{\frac{\partial C}{\partial S}}{i}\right) + \frac{C}{H} - \frac{\partial C}{\partial H}$$

$$= P - \left(\frac{\frac{\mathrm{d}G}{\mathrm{d}S}p}{i} - \frac{\frac{\partial C}{\partial S}}{i}\right) - \frac{\partial C}{\partial H}$$

$$= p - \left(\frac{\frac{\mathrm{d}G}{\mathrm{d}S}p}{i} - \frac{\frac{\partial C}{\partial S}}{i}\right) = 0 \quad (17.40)$$

which is identical to the present-value-maximising fishing rule given in equation 17.37. A tax on landed fish equal in value to the user cost of fishing would, therefore, bring about a socially efficient outcome.

Thus under conditions of open access, effort increases until price minus average cost is zero. The optimal solution is where price minus marginal cost is equal to the user cost of fishing. The tax needs to account for both the user cost of fishing and the fact that individual fishing firms under open access use average costs rather than marginal costs. This is accomplished by imposing the optimal landing tax, t, described above.[32]

In fact, there may be an additional tax adjustment required to bring about a socially efficient outcome. In the models discussed in this chapter, we have adopted simplified functions which rule out the existence of crowding diseconomies. Where crowding diseconomies do exist, these warrant a further component to the optimal tax; that component would be set at a value that internalises crowding externalities. Overall, therefore, an optimal landing tax will have three components, the first two of which are incorporated in the expression for t^* given above in expression 17.39:

1. One part corrects for the fact that fishermen in open access take no account of the future benefits to be obtained by refraining from harvesting.
2. A second part consists of the difference between average and marginal cost of fishing (at the social optimum).
3. A third part internalises crowding diseconomies.

Despite their theoretical attractiveness, tax systems of this kind are, to the authors' knowledge, non-existent. The reasons for this are likely to be that such taxes are unenforceable (or at least only enforceable at very high cost) and that they conflict with the desire to maintain fishermen's incomes.[33]

17.14.3 Incentive-based instruments: property rights and transferable harvesting quotas

A central theme of this chapter has been the key role played by property rights – or their absence – in renewable resource exploitation. Thus one way of trying to achieve efficient resource harvesting is to define and allocate exclusive property rights to the resource. Where a resource is exclusively owned, and generates for its owner or owners the full income flow attributable to that resource, owners have incentives to maximise its present value and so to take whatever investment or conservation decisions are consistent with that goal.

Many nations have extended the limits of their national jurisdictions over marine resources to 200 miles from their coasts. Is this sufficient to meet the conditions we have just described for exclusive and enforceable property rights? In most cases, that by itself is not sufficient. Even if a government were willing and able to prevent all access to its fisheries

[32] Earlier editions of this text did not have the correct expression for the optimal landing tax, by having failed to take proper account of the average versus marginal cost distinction referred to in the discussion above.

[33] In regard to the superiority of taxes or quotas over regulation, there is in the literature an implicit, but generally incorrect, assumption that the former avoid the knowledge-limitations, monitoring and enforcement problems widely recognised with the latter. A careful inspection of what would be involved even in obtaining a good estimate of an optimal landing tax, let alone monitoring and enforcing its application, suggests right away that this presumption is likely to be false in this particular instance (and no doubt in many others too).

by foreign nationals, there may still be *de facto* open access to fishing boats of the nation in question. The problem is not resolved, and clearly something additional is required.

One way of thinking what additional instrument is required is to follow up on a device we used when introducing the notion of a 'private-property fishery'. There we presented two 'scenarios'. The first envisaged a fishery consisting of a large number of sub-fisheries, each exclusively owned by one operator. The second envisaged a fishery in which enforceable property rights are collectively vested in a particular large set of fishing firms *and* in which the behaviour of those individual owners was somehow or other coordinated to be collectively rational. We did not specify, though, what either of these mechanisms might be in practice.

There are a large number of ways in which one or other of these scenarios could be achieved or, rather, mimicked. One way involves the state or regulator acting as coordinator, allocating exclusive property rights to particular quantities of harvested fish to a particular set of operators. This requires more than merely allocating quotas or allowable catches to the fleet as a whole: the allocations should be to individuals, or possibly to some small collective units each of which can be relied upon to internally coordinate its actions.

Moreover, if these exclusive property rights are to have their full worth to those to whom they are allocated, they must – like all property rights – be transferable or marketable. Bearing in mind results obtained in Chapter 6, marketability will be consistent with the objective of achieving whatever target is sought at least cost. This corresponds with the requirement we are looking for that the system in some way mimics a coordinated present-value-maximising fishery.

One scheme that has these properties is the 'individual transferable quota' (ITQ) system. This operates (approximately) in the following way for some particular stock of a controlled species. Scientists assess current and potential stock levels, and determine a maximum total allowable catch (TAC). The TAC is then divided among fishers. Each fisher can catch and land up to the amount of the quota they hold. Alternatively, some or all of the ITQs that an operator holds can be sold to others. No entitlement exists to harvest fish in the absence of holding ITQs.

To see how the ITQ system can result in the harvest of a given target quantity of fish in a cost-efficient manner, consider the following hypothetical example in which, for arithmetical simplicity only, the industry consists of just two fishing firms. The two firms differ in terms of harvesting costs, one being low-cost ($2 per tonne) and the other high-cost ($4 per tonne). A tonne of fish can be sold for $10. Each firm has historically caught and sold 100 tonnes of fish each period. Now consider what will happen if the government imposes an industry TAC of 100 tonnes. Suppose that a non-transferable quota of 50 tonnes is imposed on each firm. The total catch will be 100 tonnes, at a total cost of $300 (that is, $50 \times 4 + 50 \times 2$). Next, suppose that a transferable quota of 50 tonnes is allocated to each fisher; what will happen in this case? Given that the low-cost fisher makes a profit (net price) of $8 per tonne, while the high-cost fisher makes a profit of $6 per tonne, a mutually advantageous trade opportunity arises. Suppose that an ITQ price is agreed at $7 per tonne of landed fish. The high-cost producer will sell quotas at this price, obtaining a higher value per sold quota ($7) than the profit forgone on fish it could otherwise catch ($6). The low-cost producer will purchase ITQs at $7, as this is lower than the marginal profit ($8) it can make from the additional catch that is permitted by possession of an ITQ. A Pareto gain takes place, relative to the case where the quotas are non-transferable. This gain is a gain for the economy as a whole as can be seen by noting the total costs after ITQ trading. In this case, all 50 ITQs will be transferred, and so 100 tonnes will be harvested by the low-cost fisher, at a total cost of $200.

This example is, of course, unrealistically simple. Nevertheless, the underlying principle applies generally to marketable permit or quota systems in a wide set of circumstances, for any large number of operators, and with possibly heterogeneous and non-constant marginal harvesting costs. Transferability ensures that a market will develop in the quotas. In this market, high-cost producers will sell entitlements to harvest, and low-cost producers will purchase rights to harvest. The market price will be set at some level intermediate between the net prices or profits of the different producers. We demonstrated in Chapter 7 that this efficiency property is also shared by a tax system; indeed, a tax rate of \$7 per tonne of harvested fish would bring about an identical outcome to that described above. Convince yourself of why this is so.

The transferable quota system has been used successfully in several fisheries, including some in Canada and New Zealand. Some aspects of its practical experience are examined in Box 17.6. One matter which warrants attention is what might be labelled a social or regional equity problem that can arise from the use of an ITQ system. It is quite possible, for example, that fishermen in isolated communities will sell their ITQs, leaving particular locations entirely without a fishing industry. This appears to have been the case in Iceland's experience with the use of an ITQ regime.[34] Given that for every job at sea there may be ten jobs on land involved in maintaining and supplying the boats and processing and distributing the fish, one can understand that politicians may be reluctant to adopt ITQ systems where the transfer of ITQs might have significant regional economic impacts.

In this section we have focused on one specific form of property rights – exclusive individual private-property rights – and one particular form of policy instrument – individual transferable quotas. However, we do not wish to imply that these are the only ways of achieving efficient outcomes. In principle, many forms of property rights regime, and many forms of policy instrument, can deliver efficient outcomes. The suggestions for further reading at the end of this chapter point you to some other possibilities.

Box 17.6 The individual transferable quota system in New Zealand fisheries

In 1986, New Zealand introduced an individual transferable quota (ITQ) system for its major fisheries. The ITQ management system operates in the way we described earlier. Government scientists annually assess fish stock levels, and determine maximum total allowable catches (TAC) for controlled species. New Zealand legislation requires that the TAC levels are consistent with the stock levels that can deliver maximum sustainable yields. The TAC is then divided among fishers, with the shares being allocated on the basis of individual catches in recent years. Each fisher can catch fish up to the amount of the quota it holds, or the quotas can be sold or otherwise traded.

The ITQ system has a number of desirable properties. First, fishermen know at the start of each season the quantity of fish they are entitled to catch; this allows effort to be directed in a cost-minimising efficient manner, avoiding the mad dash for catches that characterises free-access fishery. Secondly, as a market exists in ITQs, resources should be allocated in such a way that firms with low harvesting costs undertake fishing. The reasoning behind this assertion is explained in the main text.

The ITQ system operates in conjunction with strictly enforced exclusion from the fishery of those not in possession of quotas. This access restriction generates appropriate dynamic incentives to conserve fish stocks for the future whenever the net returns of such 'investments' are sufficiently high.

The evidence of the ITQ system in operation suggests that, in comparison with alternative management regimes that might have been implemented, it has been successful both as a conservation tool and in terms of reducing the size of the uneconomically large fleets. The ITQ system has not eliminated all problems, however. The fishing industry creates continuous pressure to push TAC levels upwards, and great uncertainty remains as to the levels at which the TAC can be set without jeopardising population numbers. The ITQ system has failed to find a clear solution to the problems of by-catch – the netting of unwanted, untargeted species – and highgrading – the discarding of less valuable species or smaller-sized fish, in order to maximise the value of quotas set in terms of fish quantities.

The ITQ system now operates, to varying extents, in the fisheries of Australia, Canada, Iceland and the United States.

Source: WR (1994), chapter 10

[34] See, for example, Eythórsson (2000a and 2000b).

Summary

- Simple models, such as the logistic growth equation, can be used to describe the biological growth properties of renewable resources. However, for some species – those which exhibit critical depensation – there will be some positive level of population size below which the stock cannot be sustained (and so will eventually collapse to zero).
- The notion of a sustainable yield is a useful heuristic device for analysing resource harvesting. Resource owners are sometimes recommended to manage stocks so as to extract a maximum sustainable yield. This is only economically efficient under special circumstances.
- Open-access fisheries are characterised by conditions of individualistic competition combined with an inability of each boat owner to appropriate the gains of their investment in fish stocks. An open-access fishery is likely to be characterised by an economically excessive amount of fishing effort.
- It is important – particularly when conditions of open access prevail – to distinguish between steady-state equilibrium outcomes and dynamic adjustment paths for a renewable resource stock. In some cases equilibrium outcomes are unobtainable (and so irrelevant) because stocks are pushed beyond critical threshold points during adjustment processes.
- Open-access conditions do not *necessarily* imply stock collapses. But economic (and biological) over-harvesting is more likely to occur where the stock is exploited under conditions of open access than where access can be regulated and enforceable property rights exist.
- Comparative statics analysis (for both open-access and private-property models) suggests that steady-state stocks will be higher (fishing effort will be lower) when resource prices fall and when harvesting costs rise. Increases in the efficiency of fishing effort will reduce stocks. The effects of parameter changes on harvest quantities cannot be unambiguously signed, as harvest quantities depend on the particular configuration of stock and effort changes (which may be in opposite directions).
- Where resource owners have a positive discount rate, and aim to maximise the resource net present value (PV), resource stocks will typically be kept at smaller levels than where no discounting of cash flows is undertaken. As the discount rate becomes arbitrarily large, a PV-maximising fishery model converges in its outcomes to that of an open-access fishery.
- There are several reasons why privately rational resource exploitation decisions are not socially efficient or optimal. These include poorly defined or unenforceable property rights, a failure to internalise the value of the resource stock size (and various other kinds of harvesting external effects) in the objective function, and a divergence between private and social discount rates.
- Resource harvesting may, under certain circumstances, lead to the biological exhaustion of stocks or even extinction of species. Sometimes the stocks in question are target species; but they may also be incidental (untargeted) sufferers of harvesting behaviour.
- Outcomes in which fishery stocks are economically depleted are common in practice, as are outcomes in which stocks are being harvested beyond safe biological limits. Occasionally these processes have led to species extinction.
- Fisheries regulatory instruments have typically taken the form of some kind of total-allowable-catch licence system, combined with controls over effort, such as fishing seasons, closure of fisheries, and boat or gear restrictions. Controls have also been targeted at controlling catch ‘quality’ or composition (such as minimum mesh sizes). It is difficult to find any evidence that such instruments are effective in achieving their stated goals.

- Market-based instruments, particularly transferable fishing licences, are theoretically more attractive and have shown much promise where used to date. However, they are not a panacea for attaining fishery objectives.
- As with all environmental assets, renewable resources typically provide a multiplicity of valuable services. Their value for supply of food or materials – which drives private harvesting behaviour – is only a partial reflection of total resource values. It is not surprising, therefore, that socially optimal resource exploitation will need to use several instruments, and/or be subject to a variety of constraints.

Further reading

General reviews

Excellent reviews of the state of various renewable resources in the world economy, and experiences with various management regimes are contained in the biannual editions of *World Resources*. See, in particular, the sections in *World Resources 1994–95* on biodiversity (chapter 8) and marine fishing (chapter 10). Various editions of the *United Nations Environment Programme, Environmental Data Report* also provide good empirical accounts.

Renewable resources

Clark (1990), Conrad and Clark (1987) and Dasgupta (1982) provide graduate-level accounts, in quite mathematical form, of the theory of renewable resource depletion, as do Wilen (1985), Conrad (1995) and Rettig (1995). Good undergraduate accounts are to be found in Fisher (1981, chapter 3), the survey paper by Peterson and Fisher (1977), Neher (1990), Hartwick and Olewiler (1998) and Tietenberg (2000). Gordon (1954) is a classic paper developing the idea of open-access resources. Bromley (1999) examines common property resource regimes, and argues that many fisheries have transformed to open access with population growth. Ostrom *et al.* (2002) provides a multi-disciplinary reappraisal of the role of the notion of the commons as an explanation of human activity. Other references on open access are Ostrom (1998) and Baland and Platteau (1996), the latter of which includes a game-theoretic approach.

Fisheries

A Word document describing the current state of global fisheries is available in the *Additional Materials* as *The current state of marine fisheries.doc*. Comprehensive accounts of the world's fisheries can be found from the UN FAO at www.fao.org. In particular, see the FAO's publication *State of World Fisheries and Aquaculture*, updated every few years and available online. FAO (2002) gives a good account of recent thinking about sustainability, property rights, and the precautionary principle in fisheries management. Works which focus specifically on fisheries include the early classics by Munro (1981, 1982); Hannesson (1993); and Cunningham *et al.* (1985). Van Kooten and Bulte (2000) contains an excellent survey of the economics of fishing, covering with great care harvesting under various forms of uncertainty. Iudicello *et al.* (1999) examines the economics of overfishing. Anderson (1986) deals with fisheries management. More advanced discussions of fisheries management can be found in Conrad and Clark (1987), Harris (1998) which discusses 'high grading', Neher (1990), Graves *et al.* (1994), Salvanes and Squires (1995), Sutinen and Anderson (1985), Ludwig *et al.* (1993), Tahvonen and Kuuluvainen (1995), and L.G. Anderson (1977, 1981, 1995), the last of which examines gear restrictions and the ITQ system. F.J. Anderson (1985, chapter 7) gives a very thorough and readable analysis of policy instruments that seek to attain efficient harvesting of fish stocks, using evidence from Canadian experience. That book also provides a good account of models of fluctuating fish populations, an issue of immense practical importance. Rettig (1995) considers the problem of management with migratory and transboundary populations.

There are now several very useful Web-based sources of information on fishery problems and fishery management. One such source is the website of the International Institute of Fisheries Economics and Trade (IIFET) at http://osu.orst.edu/Dept/IIFET (search via 'Resources') which contains conference proceedings and contributions on ITQ management and many other fisheries- and resource-related topics.

Empirical models include Wilen (1976) on North Pacific fur seal; Amundsen *et al.* (1995) on North Atlantic minke whale; Bjorndal and Conrad (1987) on North Sea herring; Henderson and Tugwell (1979) on a lobster fishery; and Huppert (1990) on Alaskan groundfish.

Ecological considerations

Ecologists have a fundamentally different notion of population viability to that held by many environmental economists. In particular, they typically reject the largely deterministic bioeconomic models examined in this chapter, and would argue that ecological complexities cannot be properly dealt with by a fixed number for S_{MIN}, nor by the simple inclusion of a stochastic factor in an otherwise deterministic growth function. It is, therefore, sensible to read some expositions of issues relating to resource exploitation and conservation, and the nature of stochasticity in biological systems written from an ecological perspective. Krebs (1972, 1985) contain good expositions of ecology. MacArthur and Wilson (1967) consider spatial aspects of biological population dynamics (island biogeography; metapopulations); Lande *et al.* (1994) discuss extinction risk in fluctuating populations. The following three references all give accounts written from an ecological perspective on issues discussed in this chapter and elsewhere throughout the text.

May (1994) provides an excellent short account of the economics of extinction;
Ludwig *et al.* (1993) provide a historical perspective to resource scarcity and resource conservation; and Hilborn *et al.* (1995) investigate sustainable resource management.
Sethi and Somanathan (1996) discuss the use of evolutionary game theory to examine ecological systems.

Species extinction and biodiversity decline

The various causes of biodiversity loss are surveyed in a separate document in the *Additional Materials*, *What is causing the loss of biodiversity.doc*. A simple, non-technical account of species loss arising from harvesting and human predation is given in Conrad (1995). For a more rigorous and complete account, see Clark (1990). Barbier *et al.* (1990b) examine elephants and the ivory trade from an economics perspective. In addition to those, important ecological accounts of the issue are Lovelock (1989, developing the Gaia principle), Ehrenfeld (1988) and Ehrlich and Ehrlich (1981, 1992). The classic book in this field is Wilson (1988). More recent evidence is found in Groombridge (1992), Hawksworth (1995), Jeffries (1997) and UNEP (1995).

Perrings (1995), Jansson *et al.* (1994) and Perrings *et al.* (1995) examine biodiversity loss from an integrated ecology–economy perspective. Several other ecological accounts, together with more conventional economic analyses of the causes, are found in Swanson (1995a, 1995b) and OECD (1996). Repetto and Gillis (1988) examine biodiversity in connection with forest resource use. The economics of biodiversity is covered at an introductory level in Pearce and Moran (1994), McNeely (1988) and Barbier *et al.* (1994).

Other economics-oriented discussions are in Simon and Wildavsky (1993). There is also a journal devoted to this topic, *Biodiversity and Conservation*. Articles concerning biodiversity are regularly published in the journal *Ecological Economics*. Policy options for conserving biodiversity are covered in OECD (1996). Common and Norton (1992) study conservation in Australia.

Conservation

A large literature now exists examining the economics of wilderness conservation, including Porter (1982), Krutilla (1967) and Krutilla and Fisher (1975).

Excellent accounts of the notion of a safe minimum standard of conservation are to be found in Randall and Farmer (1995) and Bishop (1978). Other good references in this area include Ciriacy-Wantrup (1968), Norgaard (1984, 1988), Ehrenfeld (1988) and Common (1995).

Water

Water may, of course, be regarded as a renewable resource. Good discussions of the valuation of water quality improvements are found in Desvousges *et al.* (1987) and Mitchell and Carson (1984) (which both emphasise valuation issues), Ecstein (1958) and Maass *et al.* (1962), both of which focus on CBA. Water quality management is considered by Johnson and Brown (1976), Kneese (1984) and Kneese and Bower (1968). The US EPA website contains much useful information about groundwater (and drinking water) resources (web links include www.epa.gov/safewater/protect/sources.html and www.epa.gov/OGWDW). It may also be helpful to check out the site of the Groundwater Foundation (a non-profit organisation) at www.groundwater.org/GWBasics/gwbasics.htm and the US Geological Survey (USGS) groundwater information web sites (http://ga.water.usgs.gov).

Dams

Large dam construction and operations can have dramatic effects on human and biological populations. A large literature on this topic has been spawned by the World Bank Operations Evaluation Department and the IUCN–The World Conservation Union. Follow their respective websites. See also the World Commission on Dams (2000) with website at www.damsreport.org.

Land-use policy

El-Swaify and Yakowitz (1998) is a general survey.

Discussion questions

1. What fraction of the world's protein requirements is met by fish? What are the implications of your answer?
2. Would the extension of territorial limits for fishing beyond 200 miles from coastlines offer the prospect of significant improvements in the efficiency of commercial fishing?
3. Discuss the implications for the harvest rate and possible exhaustion of a renewable resource under circumstances where access to the resource is open, and property rights are not well defined.
4. To what extent do environmental 'problems' arise from the absence (or unclearly defined assignation) of property rights?
5. Fish species are sometimes classified as 'schooling' (such as herring, anchovies and tuna) or 'searching' (non-schooling) classes, with the former being defined by the tendency to 'school' in large numbers. In the text we specified fishery harvest by the equation $H = H(E, S)$. For some species, the level of stocks has a much less important effect on harvest, and so (as an approximation) we may write $H = H(E)$. Is this more plausible for schooling or searching species, and why?

Problems

Problems marked with an asterisk * require that the reader either construct his or her own spreadsheet program, or adapt the file *exploit5.xls*.

1. (a) The simple logistic growth model given as equation 17.3 in the text

$$G(S) = g\left(1 - \frac{S}{S_{\text{MAX}}}\right)S$$

gives the amount of biological growth, G, as a function of the resource stock size, S. This equation can be easily solved for $S = S(t)$, that is, the resource stock as a function of time, t. The solution may be written in the form

$$S(t) = \frac{S_{\text{MAX}}}{1 + ke^{-gt}}$$

where $k = (S_{\text{MAX}} - S_0)/S_0$ and S_0 is the initial stock size (see Clark, 1990, p. 11 for details of the solution). Sketch the relationship between $S(t)$ and t for:

(i) $S_0 > S_{\text{MAX}}$

(ii) $S_0 < S_{\text{MAX}}$

*(b) An alternative form of biological growth function is the Gompertz function

$$G(S) = gS\ln\left(\frac{S_{\text{MAX}}}{S}\right)$$

Use a spreadsheet program to compare – for given parameters g and S_{MAX} – the growth behaviour of a population under the logistic and Gompertz growth models.

2. A simple model of bioeconomic (that is, biological and economic) equilibrium in an open-access fishery in which resource growth is logistic is given by

$$G(S) = g\left(1 - \frac{S}{S_{\text{MAX}}}\right)S - eES$$

and

$$B - C = PeES - wE = 0$$

with all variables and parameters defined as in the text of the chapter.

(i) Demonstrate that the equilibrium fishing effort and equilibrium stock can be written as

$$E = \frac{g}{e}\left(1 - \frac{w}{PeS_{\text{MAX}}}\right) \text{ and } S = \frac{w}{Pe}$$

(ii) Using these expressions, show what happens to fishing effort and the stock size as the 'cost-price ratio' w/P changes. In particular, what happens to effort as this ratio becomes very large? Explain your results intuitively.

3. In what circumstances would it be plausible to assume that, as a first approximation, harvest costs do *not* depend on stock size?

4. (i) The results of this chapter have shown that the outcomes (for S, E and H) are identical in what has been called the PV-maximising model and the static private-fishery profit-maximising model when the discount rate is zero. Explain why this is so. Also explain why the stock level is higher under zero discounting than under positive discounting.

(ii) It has also been shown in this chapter that as the interest rate becomes arbitrarily large, the PV-maximising outcome converges to that found under open access. Why should this be the case? (If you are using the *exploit5.xls* spreadsheet, this result can be quickly verified. In the worksheet 'Steady states (2)', note that at $i = 1000$, the present-value outcome is more or less identical to that which emerges under open access.)

5. Calculate the 'growth rate', dG/dS, at which the fish population is growing in the open-access equilibrium, the static private-property equilibrium, and the present-value-maximising equilibrium with costs dependent on stock size and $i = 0.1$, for the baseline assumptions given in Table 17.3. At what stock size is $dG/dS = 0$ (the maximum sustainable yield harvest level)? Explain and comment on your findings.

6. Demonstrate that open access is not cost-minimising.

7. Table 17.5 demonstrated that while the signs of partial derivatives for open access and static private property were identical for the steady-state equilibrium stock and effort, for the steady-state harvest level three of the partial derivatives that could not be signed under open-access conditions can be signed under the (static) private-property case. What intuition can be used to explain this additional determinacy in the private-property case?

CHAPTER 18 Forest resources

This was the most unkindest cut of all.

William Shakespeare, Julius Caesar III.ii (188)

Learning objectives

Having completed this chapter, the reader should be able to

- understand the various functions provided by forest and other woodland resources
- describe recent historical and current trends in forestation and deforestation, and be aware of the uncertainties that relate to some forestry data
- recognise that plantation forests are renewable resources but natural – particularly primary – forests are perhaps best thought of as non-renewable resources in which development entails irreversible consequences
- explain the key differences between plantation forests and other categories of renewable resource
- understand the concepts of site value of land and land rent
- use a numerically parameterised timber growth model, in conjunction with a spreadsheet package, to calculate appropriate physical measures of timber growth and yield; and given various economic parameters, to calculate appropriate measures of cost and revenue
- obtain and interpret an expression for the present value of a single-rotation stand of timber
- using the expression for present value of a single rotation, obtain the first-order condition for maximisation of present value, and recognise that this can be interpreted as a modified Hotelling rule
- undertake comparative static analysis to show how the optimal stand age will vary with changes in relevant economic parameters such as timber prices, harvesting costs and interest (or discount) rate
- specify an expression for the present value of an infinite sequence of identical forest rotations, obtain an analytic first-order expression for maximisation of that present value with respect to the rotation age, and carry out comparative static analysis to ascertain how this varies with changes in economic parameters

Introduction

This chapter is concerned with forests and other wooded land. In the first section, the present state of global forest resources is briefly described. We then consider several salient characteristics of forest resources. This draws your attention to some of the particular characteristics of forest resources that differentiate its study from that of fisheries, the principal focus of Chapter 17.

Roughly speaking, forest resources can be divided into three categories: natural forests, semi-natural (disturbed or partly developed) forests, and plantation forests.[1] As we shall see, these are very different

[1] Here we use the word ‘forest’ to refer to both forested land and (the less densely stocked) woodland. However, in other parts of this chapter, the word ‘forest’ is used in a more restrictive sense. Where that happens, the text will define the particular meaning being applied to that term.

in terms of the services that they provide. Our attention in this chapter is largely given to the two 'extreme' cases of natural and plantation forests. Semi-natural forests are a hybrid form, and will share characteristics of the two other cases depending on the extent to which they have been disturbed or managed.

Section 18.3 considers plantation forests. The analysis of plantation forestry is well developed, and it has been the object of an important sub-discipline within economics for well over a century. A plantation forest is a renewable resource, and the techniques we outlined in the previous chapter can be applied to the analysis of it. However, the long span of time taken by trees to reach maturity means that the age at which a stand of trees is cut – the rotation period – is of central (but not exclusive) importance, and is the dimension on which our analysis focuses.

Initially, our emphasis is on the timber yielded by managed forest land. However, all forests – even pure plantation forests – provide a wide variety of other, non-timber, benefits. Forestry policy in many countries is giving increasing weight to non-timber values in forest management choices. Sections 18.4 and 18.5 investigate the question of how forests should be managed when they are used, or generate values, in multiple ways.

Not surprisingly, natural (undisturbed) forests are biologically the most diverse and perform a much broader range of ecological, amenity and recreational and other economic services than do plantation forests. We devote Section 18.6 to looking at deforestation of natural woodland. Particular attention is paid to tropical deforestation, an issue that has become the subject of extensive study within environmental economics in recent decades.

Plantation forests are renewable resources. Does the same hold for undisturbed natural forests? The fact that trees can grow and reproduce suggests that this is so. But a little reflection suggests that matters are not quite so straightforward. If we think about natural forests as ecosystems providing multiple services, and recognise that the ways in which such forests are typically 'developed' or disturbed generate irreversible changes, it becomes clear that they share some of the characteristics of non-renewable resources. Hence it may be preferable, under present conditions at least, to regard natural forests as existing in more-or-less fixed quantities and once 'mined' as being irreversibly lost as natural forests. Although trees may subsequently grow in areas once occupied by natural forest, the gestalt of what constitutes a natural forest cannot be replaced (except over extremely long spans of time). We examine these issues in Section 18.6. The chapter concludes with a brief discussion of government and forest resources.

Sections 18.2 and 18.3 make extensive use of economic models of forestry. Several illustrative examples are used in those parts of the chapter. To allow the reader to replicate our results, and to explore the properties of these models a little further, all calculations in this chapter – and all associated diagrams – are performed using Excel workbooks. *Chapter18.xls* contains the calculations and charts used in the main body of the chapter. *Palc18.xls* contains computations used in Appendix 18.2 and in some of the problems at the end of this chapter. Details of other associated Excel files are given below at appropriate places. These files can be found on the Companion Website.

18.1 The current state of world forest resources

Comprehensive assessments of the state of the world's forest resources are published every five years by the Food and Agriculture Organisation (FAO) of the United Nations in its 'Global Forest Resources Assessment' series. At the time of writing, the process for compiling the 2010 edition had begun, but the latest published version was *Global Forest Resources Assessment 2005* (FAO, 2005), labelled subsequently in this chapter as 'FRA 2005'. The complete report is available online by searching from the forestry section of the FAO website at http://www.fao.org/forestry/home/en/. Material in this section is largely drawn from that report. Readers should note that whilst UNFAO forestry and woodland data is indeed more comprehensive than that from any other source, there are legitimate concerns about its reliability. This is explored in Box 18.2 near the end of this section.

Box 18.1 Forest, other woodland, and other land with tree cover: definitions of terms

Forest	Land spanning more than 0.5 hectares with trees higher than 5 metres and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ. It does not include land that is predominantly under agricultural or urban land use.
Other wooded land	Land not classified as forest, spanning more than 0.5 hectares; with trees higher than 5 m and a canopy cover of 5–10 percent, or trees able to reach these thresholds in situ; or with a combined cover of shrubs, bushes and trees above 10 percent. It does not include land that is predominantly under agricultural or urban land use.
Other land with tree cover	All land that is **not** classified as forest or other wooded land is called '*other land*'. Of this, '*other land with tree cover*' is defined as land spanning more than 0.5 hectares with a canopy cover of more than 10 percent of trees able to reach a height of 5 m at maturity.

One can partition the earth's total land area into four categories:

1. forest land with a high density of tree cover;
2. other wooded land, that is, extensively wooded, but where the density or extensiveness of trees is insufficient to warrant description by the word 'forest';
3. other land with tree cover, which consists of land that has substantial wooded coverage, but where the density of that coverage is fairly low or the trees are relatively small;
4. other land (that does not have forest or tree coverage); this includes agricultural land, meadows and pastures, built-up areas, barren land, and land incapable of supporting large trees or trees at anything other than very low density.

Box 18.1 gives more complete definitions of these terms, as used by the United Nations FAO. Table 18.1, based on data taken from FRA 2005, gives total areas of the three categories of forest and wooded land listed in Box 18.1 for continents of the world and various sub-continental areas. The earth's total land area is a little larger than 13 thousand million hectares. Of this, approximately 5.4 thousand million ha (that is, 41%) consists of forest, other wooded land, or other land with tree cover. Of this, the first two categories are dominant, as can be seen from Table 18.1.

In discussions concerning forest and woodlands attention is given mainly to forest land, because of its overriding importance as an environmental asset and as a source of commercial and amenity values. The rest of this section will deal exclusively with forest land for those reasons. Table 18.2 shows the extent of forest land as a proportion of total land area for each of the continents and regional areas identified above, and also shows how forests are geographically distributed.[2,3] Perhaps surprisingly, Europe has the largest share of the world's forest, over 25% of the total. This is mainly a reflection of the huge extent of forests in Russia, containing over 20% of the world's total forest area. (Russia's forests are less dominant in terms of biomass, however. For example, while Russia's forest area is about two-thirds larger than that of Brazil, the biomass contained in Russian forests is one-fifth smaller than that in Brazil's forests.)

Perhaps the most commonly used indicator of the state of the world's forests is the net annual rate of change of forest area.[4] Recent values of this indicator are given in Table 18.3. It is evident that the overall change is one of falling total forest area,

[2] Classified by climate zone, of total forest area, 47% is found in the tropical zone, 9% in the sub-tropics, 11% in the temperate zone and 33% in the boreal zone.

[3] FRA 2005 points out that the global forest area in 2005 of just under 4 billion hectares (ha) corresponds to an average of 0.62 ha per capita. It goes on to say that: 'However, the area of forest is unevenly distributed. For example, 64 countries with a combined population of 2 billion have less than 0.1 ha of forest per capita. The ten most forest-rich countries account for two-thirds of total forest area. Seven countries or areas have no forest at all, and an additional 57 have forest on less than 10 percent of their total land area.'

[4] Forest area change is one of the 48 indicators of the Millennium Development Goals.

Table 18.1 Extent of forest and wooded land, 2005

Country/area	Forest 1000 ha	Other wooded land 1000 ha	Other land with tree cover 1000 ha	Total forest and wooded land area 1000 ha
Total Eastern and Southern Africa	226 534	167 023	10 345	403 902
Total Northern Africa	131 048	94 609	10 207	235 864
Total Western and Central Africa	277 829	144 468	788	423 085
Total Africa	**635 412**	**406 100**	**21 339**	**1 062 851**
Total East Asia	244 862	90 003	0	334 865
Total South and Southeast Asia	283 127	29 842	10 806	323 775
Total Western and Central Asia	43 588	71 446	1 145	116 179
Total Asia	**571 577**	**191 291**	**11 951**	**774 819**
Total Europe	**1 001 394**	**100 925**	**8 044**	**1 110 363**
Total Caribbean	5 974	1 310	339	7 623
Total Central America	22 411	5 018	449	27 878
Total North America	677 464	111 866	32 899	822 229
Total North and Central America	**705 849**	**118 194**	**33 687**	**857 730**
Total Oceania	**206 254**	**429 908**	**145**	**636 307**
Total South America	**831 540**	**129 410**	**613**	**961 563**
WORLD	**3 952 025**	**1 375 829**	**75 779**	**5 403 633**

1 hectare (ha) = 10 000 square metres (m^2) = 0.01 square kilometres (km^2)
Source: Global Forest Resources Assessment 2005

Table 18.2 Forest cover by subregion 2005 and distribution

Region/subregion	Land area (1000 ha)	Forest area, 2005 (1000 ha)	Forest area as % of region's land area	Forest area as % of global forest area
Eastern and Southern Africa	814 581	226 534	27.8	5.73
Northern Africa	1 517 682	131 048	8.6	3.32
Western and Central Africa	630 393	277 829	44.1	7.03
Total Africa	**2 962 656**	**635 412**	**21.4**	**16.08**
East Asia	1 147 756	244 862	21.3	6.2
South and Southeast Asia	848 952	283 127	33.4	7.16
Western and Central Asia	1 101 205	43 588	4	1.1
Total Asia	**3 097 913**	**571 577**	**18.5**	**14.46**
Total Europe	**2 260 180**	**1 001 394**	**44.3**	**25.34**
Caribbean	22 907	5 974	26.1	0.15
Central America	51 073	22 411	43.9	0.57
North America	2 069 930	677 464	32.7	17.14
Total North and Central America	**2 143 910**	**705 849**	**32.9**	**17.86**
Total Oceania	**849 116**	**206 254**	**24.3**	**5.22**
Total South America	**1 753 646**	**831 540**	**47.7**	**21.04**
WORLD	**13 067 421**	**3 952 025**	**30.3**	**100**

1 hectare (ha) = 10 000 square metres (m^2) = 0.01 square kilometres (km^2)
Source: Global Forest Resources Assessment 2005

with 8.9 million hectares being lost annually in net terms during the decade to 2000. But there is some indication of a slowdown in the rate of decline in the subsequent five-year period, with the annual loss falling to 7.3 million hectares between 2000 and 2005. With such large totals, absolute sizes of change can be rather hard to fix meaningfully in one's mind, even when units are defined in terms of sizes of

Table 18.3 Forest cover by subregion: 1990, 2000 and 2005 and annual rates of change

Region/subregion	Forest Land Area			Annual change rate			
	1990 1000 ha	2000 1000 ha	2005 1000 ha	1990–2000 1000 ha/yr	%	2000–2005 1000 ha/yr	%
Eastern and Southern Africa	252 354	235 047	226 534	–1 731	–0.7	–1 702	–0.7
Northern Africa	146 093	135 958	131 048	–1 013	–0.7	–982	–0.7
Western and Central Africa	300 914	284 608	277 829	–1 631	–0.6	–1 356	–0.5
Total Africa	**699 361**	**655 613**	**635 412**	**–4 375**	**–0.64**	**–4 040**	**–0.62**
East Asia	208 155	225 663	244 862	1 751	0.8	3 840	1.6
South and Southeast Asia	323 156	297 380	283 127	–2 578	–0.8	–2 851	–1.0
Western and Central Asia	43 176	43 519	43 588	34	0.1	14	n.s.
Total Asia	**574 487**	**566 562**	**571 577**	**–792**	**–0.14**	**1 003**	**0.18**
Total Europe	**989 320**	**998 091**	**1 001 394**	**877**	**0.09**	**661**	**0.07**
Caribbean	5 350	5 706	5 974	36	0.6	54	0.9
Central America	27 639	23 837	22 411	–380	–1.5	–285	–1.2
North America	677 801	677 971	677 464	17	n.s.	–101	n.s.
Total North and Central America	**710 790**	**707 514**	**705 849**	**–328**	**–0.05**	**–333**	**–0.05**
Total Oceania	**212 514**	**208 034**	**206 254**	**–448**	**–0.21**	**–356**	**–0.17**
Total South America	**890 818**	**852 796**	**831 540**	**–3 802**	**–0.44**	**–4 251**	**–0.50**
WORLD	**4 077 291**	**3 988 610**	**3 952 025**	**–8 868**	**–0.22**	**–7 317**	**–0.18**

1 hectare (ha) = 10 000 square metres (m^2) = 0.01 square kilometres (km^2)
Source: Global Forest Resources Assessment 2005

particular countries.[5] In percentage terms (which do not suffer from such scaling difficulties), the loss of global forest area fell from 0.22% to 0.18% annually, a small change but nonetheless some cause for optimism about the future. The figures referred to earlier in this paragraph are net changes. Lying behind those net changes are gross flows decreasing and increasing the stocks. For example, FRA 2005 statistics show that 'deforestation, mainly conversion of forests to agricultural land, continues at an alarmingly high rate – about 13 million hectares per year. At the same time, forest planting, landscape restoration and natural expansion of forests have significantly reduced the net loss of forest area.'

Looking at particular regions, it can be seen from Table 18.3 that South America and Africa suffered the largest net loss of forests in the period 2000 to 2005 – about 4.3 million and 4.0 million hectares per year respectively. North and Central America and Oceania each showed relatively small annual net losses. In contrast, Asia as a whole turned round net annual losses of 800 000 ha in the 1990s to a net gain of 1 million hectares per year from 2000 to 2005, primarily as a result of large-scale afforestation reported by China. Europe's forest areas continued to grow in net terms, although more slowly than during the decade to 2000.

FRA 2005 also gives its readers some indication of the makeup of total forest area by forest type. The percentage breakdown is as follows:

2005 Total forest area	3 952 025 000 ha
of which:	(%)
Primary forest	36.4
Modified natural forest	52.7
Semi-natural forest	7.1
Productive forest plantation	3
Protective forest plantation	0.8

Natural or semi-natural forests are the predominant form forest area, accounting for 96.2% of the total. The remaining 3.8% consists of plantation forest land. The former is typically not managed at all (and where it is managed, is not done so primarily for timber production), whereas plantations are commercially operated resources, managed predominantly for timber revenues.

5 For example, Austria has a land area of about 8.3 million hectares; Panama an area of about 7.4 million. These are roughly comparable to the annual losses we are referring to here.

More than one-third of all forests are primary forests – forests of native species, in which there are no clearly visible indications of human activity and ecological processes are not significantly disturbed. Unfortunately, the area of primary forests has been falling rapidly for many years (at an average of about 6 million hectares annually). Some of this loss is due to straightforward deforestation; but also of significance are selective logging and human intrusion, both of which convert primary forests into modified natural forests. That primary forests are hugely important sources of diverse biological material, and play important roles in both local and global climates, goes a long way to explain the international concern about the absence of any sign of this rate of loss declining.

For reasons that will be explained below, a substantial part of this chapter deals with plantation forestry, even though it accounts for less than 5% of total forest area. There are two principal purposes for the establishment of plantations. The first is wood and fibre production, in what are called 'productive' forest plantations. Approximately 35% of all roundwood – all wood in the rough, for both domestic and industrial purposes – is derived from plantations. Moreover, the expansion of plantations has important effects on fuelwood availability, reducing the pressures on natural forests to provide this resource.

A second purpose relates to a variety of 'protective' roles played by plantation forests, particularly conservation of soil and water. We shall have more to say about the protective roles of forest land later in this chapter. While the proportion of plantation forests in total forest land is small it is growing quickly, at an average of 2.8 million ha per year during the period 2000–2005, with approximately half on land previously under non-forestry use.

In summary, the major changes affecting the world's forests in the period from 1990 until 2005 are:

- a large loss in tropical forest cover with a much smaller gain in non-tropical forest area;
- a large loss in natural forest area with a much smaller gain in forest plantation area;
- for the broad aggregates considered here, a loss in total forest area in all regions except Asia and Europe;
- deforestation continues at an alarmingly high rate, but the net loss of forest area is slowing down thanks to forest planting, landscape restoration and natural expansion of forests on abandoned land.

Box 18.2 The reliability of UN FAO forestry and woodland data

UN FAO forestry data is compiled from individual country estimates produced by national forestry departments. Statistics compiled on the basis of self-reporting by national forestry departments are not always reliable, particularly where the extent of forest is immense, or the resources devoted to forestry administration are scarce relative to the demands being placed on them, or where policing and monitoring of land-use changes are poor. In such cases, there is legitimate concern that forestry civil servants cannot know with accuracy the extent of illegal and unreported logging or forestry conversion, biasing upwards reported figures on areas of forested land. Hence, the national data on which FAO estimates are compiled are neither independent nor scientifically validated. Estimates of rates of forest loss are often very different from studies that use remote sensing and satellite imaging techniques.

Critics of FAO data have also questioned the appropriateness of its definitions. As you saw in Table 18.1, one of the FAO criteria used to classify forest is that tree coverage is at least 10% of total land area. But on this basis much of the world's savanna land is treated as 'forest'. Moreover, an area which was once densely forested can experience a large degree of felling or heavy degradation and still be classified as forest. Critics argue that even if accurate records existed, FAO measures of forestry will tend to be unresponsive to significant land-use changes, and can be upwardly biased over long periods of time. Box 18.3 later in this chapter gives some flavour of the degree of uncertainty that currently exists concerning rates of rainforest deforestation.

18.2 Characteristics of forest resources

Let us begin by summarising some of the key characteristics of forest resources, noting several similarities and differences between forest and marine resources (the latter being main subject of the previous chapter).

1. While fisheries typically provide a single service, forests are multi-functional. They directly provide timber, fuelwood, food, water for drinking and irrigation, stocks of genetic resources, and other forest products. Moreover, as ecosystems, forests also provide a wide variety of services, including removal of air pollution, regulation of atmospheric quality, nutrient cycling, soil creation, habitats for humans and wildlife, watershed maintenance, recreational facilities and aesthetic and other amenities. Because of the wide variety of functions that forests perform, timber managed for any single purpose generates a large number of important external effects. We would expect that the management of woodland resources is often economically inefficient because of the presence of these external effects.
2. Woodlands are capital assets that are intrinsically productive. In this, they are no different from fisheries, and so the techniques we developed earlier for analysing efficient and optimal exploitation should also be applicable (albeit with amendments) to the case of forest resources.
3. Trees typically exhibit very long lags between the date at which they are planted and the date at which they attain biological maturity. A tree may take more than a century to reach its maximum size. The length of time between planting and harvesting is usually at least 25 years, and can be as large as 100 years. This is considerably longer than for most species of fish, but not greatly different from some large animals.
4. Unlike fisheries, tree harvesting does not involve a regular cut of the incremental growth. Forests, or parts of forests, are usually felled in their entirety. It is possible, however, to practise a form of forestry in which individual trees are selectively cut. Indeed, this practice was once common, and is now again becoming increasingly common, particularly where public pressure to manage forests in a multiple-use way is strong. This form of felling is similar to the 'ideal' form of commercial fishing in which adult fish are taken, leaving smaller, immature fish unharvested for a later catch.
5. Plantation forestry is intrinsically more controllable than commercial marine fishing. Tree populations do not migrate spatially, and population growth dynamics are simpler, with less interdependence among species and less dependence on relatively subtle changes in environmental conditions.
6. Trees occupy potentially valuable land. The land taken up in forestry often has an opportunity cost. This distinguishes woodlands from both ocean fisheries (where the ocean space inhabited by fish stocks usually has no value other than as a source of fish and other marine products) and mineral deposits (where the space occupied by deposits often has little or no economic value).
7. The growth in volume or mass of a single stand of timber, planted at one point in time, resembles that illustrated for fish populations in the previous chapter.

To illustrate the assertion made in point 7, we make use of some data reported in Clawson (1977). This refers to the volume of timber in a stand of US Northwest Pacific region Douglas firs. Let S denote the volume, in cubic feet, of standing timber and t the age in years of the stand since planting. (For simplicity, we shall use a year to denote a unit of time.) The age–volume relationship estimated by Clawson for a typical single stand is the cubic function of time

$$S = 40t + 3.1t^2 - 0.016t^3$$

Figure 18.1(a) plots the volume of timber over a period up to 145 years after planting. The volume data is listed in the second column in Table 18.4. It is evident from the diagram that an early phase of slow growth in volume is followed by a period of rapid volume growth, after which a third phase of slow growth takes place as the stand moves towards maturity. The stand becomes biologically mature

Table 18.4 Present values of revenues, costs and net benefits undiscounted and discounted at 3%

Age of stand t years	Volume of timber S (cu. ft.)	Annual growth $S_t - S_{t-1}$	Interest rate $i = 0.00$			Interest rate $i = 0.03$		
			Revenue R1	Cost C1	Net benefit NB1	Revenue R2	Cost C2	Net benefit NB2
1	43.1	43.1	430.8	5 086.2	–4 655.3	418.1	5083.6	–4665.5
5	275.5	66.9	2 755.0	5 551.0	–2 796.0	2 371.3	5474.3	–3103.0
10	694.0	94.6	6 940.0	6 388.0	552.0	5 141.3	6028.3	–887.0
15	1 243.5	119.8	12 435.0	7 487.0	4 948.0	7 928.9	6585.8	1343.1
20	1 912.0	142.6	19 120.0	8 824.0	10 296.0	10 493.3	7098.7	3394.6
24	2 524.4	159.2	25 244.2	10 048.8	15 195.3	12 287.6	7457.5	4830.1
25	2 687.5	163.1	26 875.0	10 375.0	16 500.0	12 694.8	7539.0	5155.9
27	3 025.0	170.6	30 249.7	11 050.0	19 199.8	13 456.8	7691.4	5765.5
30	3 558.0	181.1	35 580.0	12 116.0	23 464.0	14 465.8	7893.2	6572.6
32	3 930.1	187.7	39 301.1	12 860.2	26 441.0	15 048.1	8009.6	7038.5
35	4 511.5	196.8	45 115.0	14 023.0	31 092.0	15 787.4	8157.5	7630.0
39	5 326.0	207.5	53 260.0	15 652.0	37 608.0	16 530.1	8306.0	8224.1
40	5 536.0	210.0	55 360.0	16 072.0	39 288.0	16 674.1	8334.8	8339.3
50	7 750.0	229.3	77 500.0	20 500.0	57 000.0	17 292.6	8458.5	8834.1
60	10 104.0	239.0	101 040.0	25 208.0	75 832.0	16 701.8	8340.4	8361.4
65	11 303.5	240.2	113 035.0	27 607.0	85 428.0	16 082.0	8216.4	7865.6
70	12 502.0	239.0	125 020.0	30 004.0	95 016.0	15 309.5	8061.9	7247.6
80	14 848.0	229.5	148 480.0	34 996.0	113 784.0	13 469.8	7694.0	5775.8
90	17 046.0	210.4	170 460.0	39 092.0	131 368.0	11 455.9	7291.2	4164.7
100	19 000.0	181.7	190 000.0	43 000.0	147 000.0	9 459.5	6891.9	2567.6
110	20 614.0	143.4	206 140.0	46 228.0	159 912.0	7 603.1	6520.7	1082.5
120	21 792.0	95.4	217 920.0	48 584.0	169 336.0	5 954.4	6190.9	–236.5
130	22 438.0	37.9	224 380.0	49 876.0	174 504.0	4 541.9	5908.4	–1366.5
135	22 531.5	5.6	225 315.0	50 063.0	175 252.0	3 925.5	5785.1	–1859.6
140	22 456.0	–29.2	224 560.0	49 912.0	174 648.0	3 367.4	5673.5	–2305.1
145	22 199.5	–66.4	221 995.0	49 399.0	172 596.0	2 865.2	5573.1	–2707.8

(reaches maximum volume with zero net growth) at approximately 135 years.[6]

How does the amount of annual growth vary with the volume of timber, S? The amount of growth is listed in the third column of Table 18.4, and the growth–volume relationship is plotted in Figure 18.1(b).[7] Although the biological growth function is not logistic in this case, it is very similar in form to simple logistic growth, being a quadratic function (with an inverted U-shaped profile).

Inspection of Figure 18.1(b) or Table 18.4 shows that the biological growth function for one stand reaches a peak current annual increment of 240 cubic feet 65 years after planting at a total standing-timber volume of approximately 11 300 cubic feet.[8] When discussing a fishery, we labelled the periodic increment at which the growth function is maximised the 'maximum sustainable yield'. But for a stand of trees all planted at one point in time, the concept of a sustainable yield of timber is not meaningful (except for specialised activities such as coppicing). While one can conceive of harvesting mature fish while leaving younger fish to grow to maturity, this cannot happen on a continuous basis in a single-aged forest stand. However, when there are many stands of trees of different ages, it is meaningful to talk about sustainable yields. This is something we shall discuss later.

[6] Inspection of Clawson's estimated timber growth equation shows that growth becomes negative after (approximately) 135 years. The equation should be regarded as being a valid representation of the growth process only over the domain $t = 0$ to $t = 135$.

[7] Table 18.4 (in a more complete form) and Figures 18.1 (a) and (b) are all generated in the Excel workbook *Chapter 18.xls*.

[8] The current annual increment (CAI) for a tree (or a stand, or a forest) is the amount of growth (in volume, diameter, or other chosen measure) over a period of one year at any stage in the tree's history. The age of the trees at the time to which the CAI refers should be specified. The mean annual increment (MAI) – or mean annual growth – refers to the average growth per year a tree or stand of trees has exhibited/experienced to a specified age. The specific age should be given when quoting MAI figures.

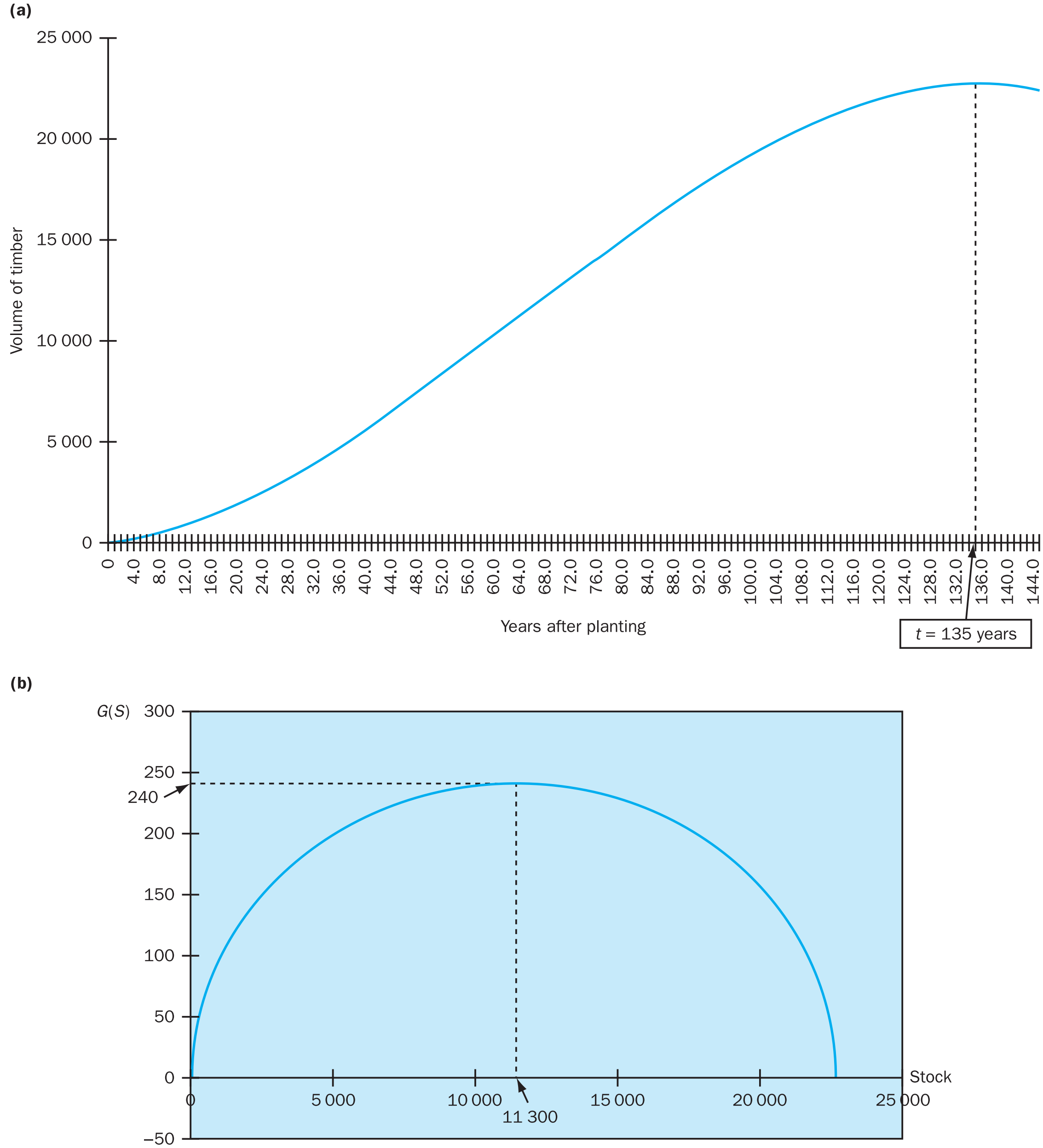


Figure 18.1 (a) The volume of timber in a single stand over time; (b) Biological growth of a single stand of timber

18.3 Commercial plantation forestry

There is a huge literature dealing with efficient timber extraction. We attempt to do no more than present a flavour of some basic results, and refer the reader to specialist sources of further reading at the end of the chapter. An economist derives the criterion for an efficient forest management and felling programme by trying to answer the following question:

What harvest programme is required in order that the present value of the profits from the stand of timber is maximised?

The particular aspect of this question that has most preoccupied forestry economists is the appropriate time after planting at which the forest should be felled. As always in economic analysis, the answer one gets to any question depends on what model is being used. We begin with one of the most simple forest models, the single-rotation commercial forest model. Despite its lack of realism, this model offers useful insights into the economics of timber harvesting. However, as we shall see later in the chapter, that which is privately optimal may not be socially efficient. In particular, where private costs and benefits fail to match their social counterparts, a wedge may be driven between privately and socially efficient behaviour. For the moment, we put these considerations to one side.

18.3.1 A single-rotation forest model

Suppose there is a stand of timber of uniform type and age. All trees in the stand were planted at the same time, and are to be cut at one point in time. Once felled, the forest will not be replanted. So only one cycle or rotation – plant, grow, cut – is envisaged. For simplicity, we also assume that

- the land has no alternative uses so its opportunity cost is zero;
- planting costs (k), marginal harvesting costs (c) and the gross price of felled timber (P) are constant in real terms over time;
- the forest generates value only through the timber it produces, and its existence (or felling) has no external effects.

Looked at from the point of view of the forest owner (which, for simplicity, we take to be the same as the landowner), what is the optimum time at which to fell the trees? The answer is obtained by choosing the age at which the present value of profits from the stand of timber is maximised. Profits from felling the stand at a particular age of trees are given by the value of felled timber less the planting and harvesting costs. Because we are assuming the land has no other uses, the opportunity cost of the land is zero and so does not enter this calculation. If the forest is clear-cut at age T, then the present value of profit is

$$(P - c)S_T e^{-iT} - k = pS_T e^{-iT} - k \tag{18.1}$$

where S_T denotes the volume of timber available for harvest at time T, i is the private consumption discount rate (which we suppose is equal to the opportunity cost of capital to the forestry firm), and lower-case p is the net price of the harvested timber. The net price of harvested timber can also be thought of as rent, as in the cases of other renewable resources (such as fish) and non-renewable resources (such as oil). In the forestry literature, this net price or rent is often referred to as stumpage. In the same vein, given that S_T denotes the volume of timber available for harvest at time T, then $(P - c)S_T$, or equivalently pS_T is what is commonly called stumpage value, the value of the proceeds from felled timber net of planting and harvesting costs.

The present value of profits is maximised at that value of T which gives the highest value for $pS_T e^{-iT} - k$. To maximise this quantity, we differentiate equation 18.1 with respect to T, using the product rule, set the derivative equal to zero and solve for T:[9]

$$\frac{d}{dT}(pS_T e^{-iT} - k) = \frac{d}{dT}(pS_T e^{-iT})$$
$$= pe^{-iT}\frac{dS}{dT} + pS^T\frac{de^{-iT}}{dT}$$

which, setting equal to zero, implies that

$$pe^{-iT}\frac{dS}{dT} - ipS_T e^{-iT} = 0$$

and so

$$p\frac{dS}{dT} = ipS_T$$

[9] Note from the first of these steps that k does not enter the first derivative, and so immediately we find that in a single-rotation model, planting costs have no effect on the efficient rotation length (provided that k is not so large as to make the maximised present value negative).

or

$$i = \frac{p\dfrac{dS}{dT}}{pS_T} \tag{18.2}$$

Equation 18.2 states that the present value of profits is maximised when the rate of growth of the (undiscounted) net value of the resource stock is equal to the private discount rate. Note that with the timber price and harvesting cost constant, this can also be expressed as an equality between the proportionate rate of growth of the volume of timber and the discount rate. That is,

$$i = \frac{\dfrac{dS}{dT}}{S_T} \tag{18.2$'$}$$

Thinking back to earlier chapters, it is evident that Equation 18.2 is an identical first-order condition to that which applies to oil and other non-renewable resources. In both cases, two rates of return are equalised. But in the case of forest (and other renewable) resources the rate of return on the right-hand side of the equation derives entirely from physical growth of the stock. This is shown explicitly in Equation 18.2′.

We can calculate the optimal, present-value maximising age of the stand for the illustrative data in Table 18.4. These calculations, together with the construction of the associated graphs are reproduced in the Excel workbook *Chapter18.xls* which can be downloaded from the Companion website. In these calculations, we assume that the market price per cubic foot of felled timber is £10, total planting costs are £5000, incurred immediately the stand is established, and harvesting costs are £2 per cubic foot, incurred at whatever time the forest is felled. The columns labelled R1, C1 and NB1 list the present values of revenues and costs and profits (labelled Net benefit in the table) for a discount rate of zero. Note that when $i = 0$, present values are identical to undiscounted values. The level of the present value of profits (NB1) over time is shown in Figure 18.2. Net benefits are maximised at 135 years, the point at which the biological growth of the stand (dS/dt) becomes zero. With no discounting and fixed timber prices, the profile of net value growth of the timber is identical to the profile of net volume growth of the timber, as can be seen by comparing Figures 18.1(a) and 18.2.

It is also useful to look at this problem in another way. The interest rate to a forest owner is the opportunity cost of the capital tied up in the growing timber stand. When the interest rate is zero, that opportunity cost is zero. It will, therefore, be in the interests of the owner to not harvest the stand as long as the volume (and value) growth is positive, which it is up to an age of 135 years. Indeed, inspection of equation 18.2′ confirms this; given that S is positive, when $i = 0$ dS/dt must be zero to satisfy the first-order maximising condition.

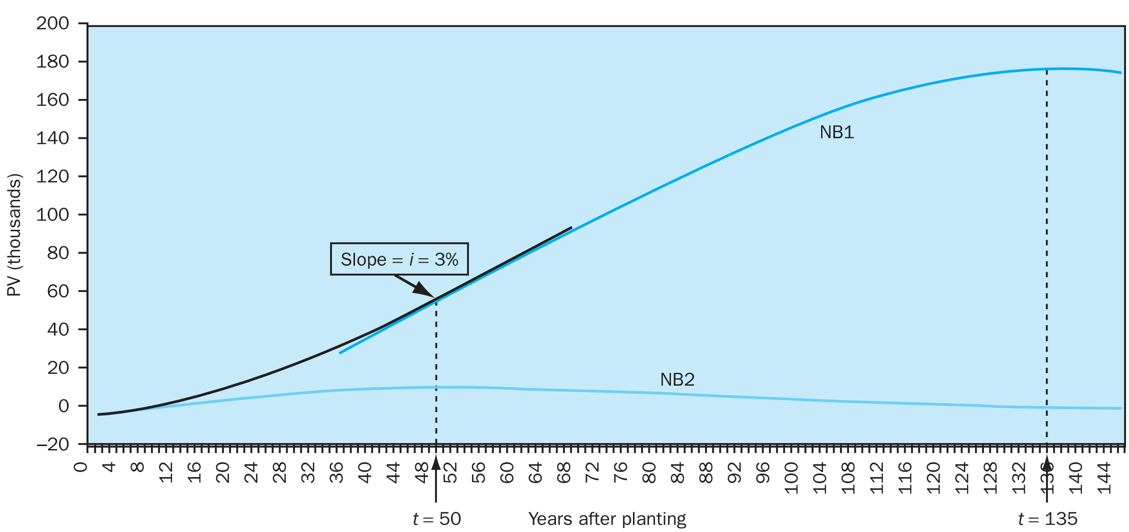


Figure 18.2 Present values of net benefits at $i = 0.00$ (NB1) and $i = 0.03$ (NB2)

Now consider the case where the discount rate is 3%. The columns labelled R2, C2 and NB2 in Table 18.4 refer to the present values of revenues, costs and profits when the interest rate is 3%. The present value of profits at a discount rate of 3% is also plotted in Figure 18.2, under the legend NB2. With a 3% discount rate, the present value of the forest is maximised at a stand age of 50 years.

Expressed in a way that conforms to equation 18.2, the growth of *undiscounted* profits,

$$\frac{p\dfrac{\mathrm{d}S}{\mathrm{d}T}}{pS_T}$$

equals i (at 3%) in year 50, having been larger than 3% before year 50 and less than 3% thereafter. This is shown by the '$i = 3\%$' line which has an identical slope to that of the NB1 curve at $t = 50$ in Figure 18.2. At that point, the growth rate of undiscounted timber value equals the interest rate. A wealth-maximising owner should harvest the timber when the stand is of age 50 years – up to that point, the return from the forest is above the interest rate, and beyond that point the return to the forest is less than the interest rate.

The single-rotation model we have used shows that the optimal time for felling will depend upon the discount rate used. It can be seen from our calculations that this effect can be huge. A rise in the discount rate from zero to 3% not only dramatically lowers the profitability of the forest but also significantly changes the shape of the present-value profile, reducing the age at which the forest should be felled (in our illustrative example) from 135 to 50 years.

More generally, it is clear from our previous arguments that as the interest rate rises the age at which the stand is felled will have to be lowered in order to bring about equality between the rate of change of undiscounted net benefits and the discount rate. In Figure 18.3, we illustrate how the optimal felling age varies with the interest rate for our illustrative data. While the exact relationship shown is only valid under the assumptions used here, it does suggest that small changes in interest rates might dramatically alter privately optimal harvesting programmes. Note that for the parameters used in this simulation, no rotation period yields a positive forest net present value above $i = 5.7\%$ and so no forest would be maintained at interest rates greater than that value. This is indicated by the horizontal line at that interest rate in the diagram.

18.3.2 Infinite-rotation forestry models

The forestry model we investigated in the previous section is unsatisfactory in a number of ways. In particular, it is hard to see how it would be meaningful

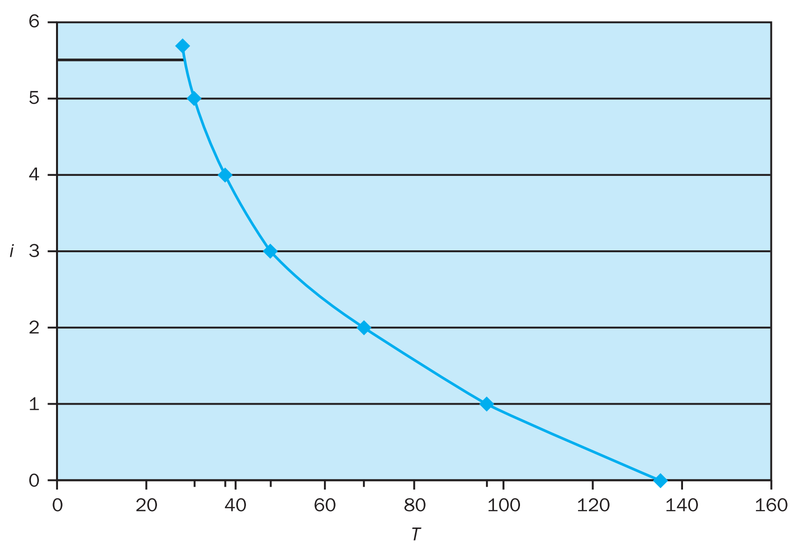


Figure 18.3 The variation of the optimal felling age with the interest rate, for a single-rotation forest

to have only a single rotation under the assumption that there is no alternative use of the land. If price and cost conditions warranted one cycle then surely, after felling the stand, a rational owner would consider further planting cycles if the land had no other uses? So the next step is to move to a model in which more than one cycle or rotation occurs. The conventional practice in forestry economics is to analyse harvesting behaviour in an infinite time horizon model (in which there will be an indefinite quantity of rotations). A central question investigated here is what will be the optimal length of each rotation (that is, the time between one planting and the next).

When the harvesting of one stand of timber is to be followed by the establishment of another, an additional element enters into the calculations. In choosing an optimal rotation period, a decision to defer harvesting incurs an additional cost over that in the previous model. We have already taken account of the fact that a delay in harvesting has an opportunity cost in the form of interest forgone on the (delayed) revenues from harvesting. But a second kind of opportunity cost now enters into the calculus. This arises from the delay in establishing the next and all subsequent planting cycles. Timber that would have been growing in subsequent cycles will be planted later. So an optimal harvesting and replanting programme must equate the benefits of deferring harvesting – the rate of growth of the undiscounted net benefit of the present timber stand – with the costs of deferring that planting – the interest that could have been earned from timber revenues and the return lost from the delay in establishing subsequent plantings.

Our first task is to construct the present-value-of-profits function to be maximised for the infinite-rotation model. We continue to make several simplifying assumptions that were used in the single-rotation model: namely, the total planting cost, k, the gross price of timber, P, and the harvesting cost of a unit of timber, c, are constant through time. Given this, the net price of timber $p = P - c$ will also be constant.

Turning now to the rotations, we assume that the first rotation begins with the planting of a forest on bare land at time t_0. Next, we define an infinite sequence of points in time that are ends of the successive rotations, t_1, t_2, t_3, ... At each of these times, the forest will be clear-felled and then immediately replanted for the next cycle. The net present value of profit from the first rotation is

$$pS_{(t_1-t_0)}e^{-i(t_1-t_0)} - k$$

that is, the volume of timber growth between the start and end of the cycle multiplied by the discounted net price of a unit of timber, less the forest planting cost. Notice that because the planting cost is incurred at the start of the rotation, no discounting is required to bring it into present-value terms. But as the timber is felled at the end of the rotation (t_1), the timber revenue has to be discounted back to its present (t_0) value equivalent.

The net present value of profits over this infinite sequence is given by

$$\begin{aligned}\Pi = {} & [pS_{(t_1-t_0)}e^{-i(t_1-t_0)} - k] \\ & + e^{-i(t_1-t_0)}[pS_{(t_2-t_1)}e^{-i(t_2-t_1)} - k] \\ & + e^{-i(t_2-t_0)}[pS_{(t_3-t_2)}e^{-i(t_3-t_2)} - k] \\ & + e^{-i(t_3-t_0)}[pS_{(t_4-t_3)}e^{-i(t_4-t_3)} - k] \\ & + \dots \end{aligned} \tag{18.3}$$

Reading this, we see that the present value of profits from the infinite sequence of rotations is equal to the sum of the present values of the profit from each of the individual rotations.

Provided conditions remain constant through time, the optimal length of any rotation will be the same as the optimal length of any other. Call the interval of time in this optimal rotation T. Then we can rewrite the present-value function as

$$\begin{aligned}\Pi = {} & [pS_T e^{-iT} - k] \\ & + e^{-iT}[pS_T e^{-iT} - k] \\ & + e^{-2iT}[pS_T e^{-iT} - k] \\ & + e^{-3iT}[pS_T e^{-iT} - k] \\ & + \dots \end{aligned} \tag{18.4}$$

Next, factorise out e^{-iT} from the second term on the right-hand side of equation 18.4 onwards to give

$$\begin{aligned}\Pi = {} & [pS_T e^{-iT} - k] + e^{-iT}\{[pS_T e^{-iT} - k] \\ & + e^{-iT}[pS_T e^{-iT} - k] \\ & + e^{-2iT}[pS_T e^{-iT} - k] + \dots\} \end{aligned} \tag{18.5}$$

Now look at the term in braces on the right-hand side of equation 18.5. This is identical to Π in equation 18.4. Therefore, we can rewrite equation 18.5 as

$$\Pi = [pS_T e^{-iT} - k] + e^{-iT}\Pi \tag{18.6}$$

which on solving for Π gives[10]

$$\Pi = \frac{pS_T e^{-iT} - k}{1 - e^{-iT}} \tag{18.7}$$

Equation 18.7 gives the present value of profits for any rotation length, T, given values of p, k, i and the timber growth function $S = S(t)$. The wealth-maximising forest owner selects that value of T which maximises the present value of profits. For the illustrative data in Table 18.4, we have used a spreadsheet program to numerically calculate the present-value-maximising rotation intervals for different values of the discount rate. (The spreadsheet is available on the Companion Website as *Chapter18.xls*, Sheet 2.) Present values were obtained by substituting the assumed values of p, k and i into equation 18.7, and using the spreadsheet to calculate the value of Π for each possible rotation length, using Clawson's timber growth equation. The results of this exercise are presented in Table 18.5 (along with the optimal rotation lengths for a single rotation forest, for comparison). Discount rates of 6% or higher result in negative present values at any rotation, and the asterisked rotation periods shown are those which minimise present-value losses; commercial forestry would be abandoned at those rates. With our illustrative data, at any discount rate which yields a positive net present value for the forest the optimal rotation interval in an infinite-rotation forest is lower than the age at which a forest would be felled in a single rotation model. For example, with a 3% discount rate, the optimal rotation interval in an infinite sequence of rotations is 40 years, substantially less than the 50-year harvest age in a single rotation. We will explain why this is so shortly.

It is also useful to think about the optimal rotation interval analytically, as this will enable us to obtain some important comparative statics results. Let us proceed as was done in the section on single-rotation forestry. The optimal value of T will be that which maximises the present value of the forest over an infinite sequence of planting cycles. To find the optimal value of T, we obtain the first derivative of Π with respect to T, set this derivative equal to zero, and solve the resulting equation for the optimal rotation length.

Table 18.5 Optimal rotation intervals for various discount rates

i	Optimal T (years) in infinite-rotation model	Optimal T (years) in single-rotation model
0	99	135
1	71	98
2	51	68
3	40	50
4	33	38
5	29	31
6	26*	26*
7	24*	22*
8	22*	19*
9	21*	17*
10	20*	15*

Notes to table:
1. Data simulated by Excel, using workbook *Chapter18.xls*
2. * For both single- and infinite-rotation models, at interest rates of 6% and above (for the price, cost and growth data used here) the PV is negative even at optimal T, so the land would not be used for commercial forestry. The value of T shown in these cases is that which minimises the PV loss.
3. To simulate the solution for $i = 0$, we used a value of i sufficiently close to (although not exactly equal to) zero so that the optimal rotation, in units of years, was unaffected by a further reduction in the value of i.

The algebra here is simple but tedious, and so we have placed it in Appendix 18.1. Two forms of the resulting first-order condition are particularly useful, each being a version of the Faustmann rule (derived by the German capital theorist Martin Faustmann in 1849; see Faustmann (1968)). The first is given by

$$\frac{p\dfrac{dS}{dT}}{pS_T - k} = \frac{i}{1 - e^{-iT}} \tag{18.8a}$$

[10] A more elegant method of obtaining equation 18.7 from 18.4 is as follows. Equation 18.4 may be rewritten as

$$\Pi = (pS_T e^{-iT} - k)\,(1 + e^{-iT} + (e^{-iT})^2 + (e^{-iT})^3 + \dots)$$

The final term in parentheses is the sum of an infinite geometric progression. Given the values that i and T may take, this is a convergent sum. Then, using the result for such a sum, that term can be written as $1/(1 - e^{-iT})$, and so

$$\Pi = \frac{pS_T e^{-iT} - k}{1 - e^{-iT}}$$

and the second, after some rearrangement of 18.8a, is given by

$$p\frac{dS_T}{dT} = ipS_T + i\Pi \tag{18.8b}$$

Problem 6 at the end of this chapter invites you to compare the first-order condition given by equation 18.8 with the equivalent first-order conditions for oil extraction and for harvesting of fish.

Either version of equation 18.8 is an efficiency condition for present-value-maximising forestry, and implicitly determines the optimal rotation length for an infinite rotation model in which prices and costs are constant.[11] Given knowledge of the function $S = S(t)$, and values of p, i and k, one could deduce which value of T satisfies equation 18.8 (assuming the solution is unique, which it usually will be). The term Π in equation 18.8b is called the site value of the land – the capital value of the land on which the forest is located. This site value is equal to the maximised present value of an endless number of stands of timber that could be grown on that land.

The two versions of the Faustmann rule offer different advantages in helping us to make sense of optimal forest choices. Equation 18.8b gives some intuition for the choice of rotation period. The left-hand side is the increase in the net value of the timber left growing for an additional period. The right-hand side is the value of the opportunity cost of this choice, which consists of the interest forgone on the capital tied up in the growing timber (the first term on the right-hand side) and the interest forgone by not selling the land at its current site value (the second term on the right-hand side). An efficient choice equates the values of these marginal costs and benefits. More precisely, equation 18.8b is a form of Hotelling *dynamic* efficiency condition for the harvesting of timber. This is seen more clearly by rewriting the equation in the form:

$$\frac{p\left(\dfrac{dS}{dT}\right)}{pS_T} = i + \frac{i\Pi}{pS_T} \tag{18.9}$$

Equation 18.9 states that, with an optimal rotation interval, the proportionate rate of return on the growing timber (the term on the left-hand side) is equal to the rate of interest that could be earned on the capital tied up in the growing timber (the first term on the right-hand side) plus the interest that could be earned on the capital tied up in the site value of the land ($i\Pi$) expressed as a proportion of the value of the growing timber (pS_T).

We can use the other version of the Faustmann rule – equation 18.8a – to illustrate graphically how the optimal rotation length is determined. This is shown in Figure 18.4. The curves labelled 0%, 1%, 2% and 3% plot the right-hand side of equation 18.8a for these rates of interest. The other, more

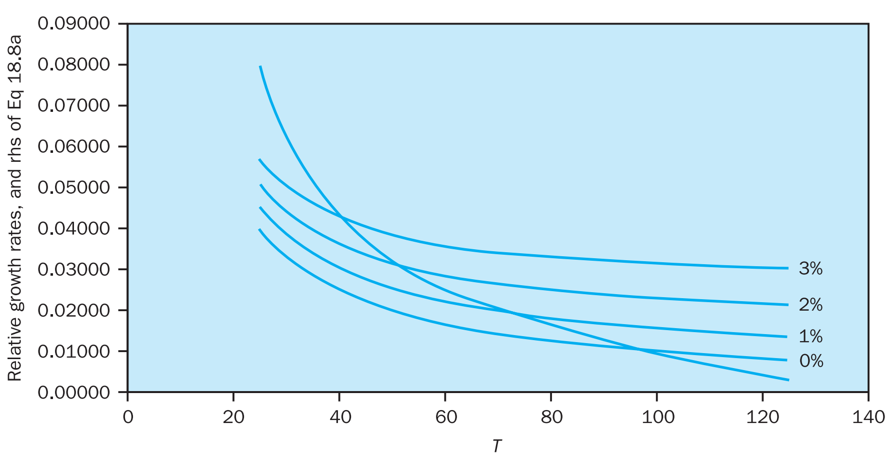


Figure 18.4 Optimal rotation lengths, *T*, as determined by equation 18.8a

[11] Unlike in the case of a single-rotation model, planting costs *k* do enter the first derivative. So in an infinite-rotation model, planting costs do affect the efficient rotation length.

steeply sloped, curve plots the left-hand side of the equation. At any given interest rate, the intersection of the functions gives the optimum T. The calculations required to generate Figure 18.4 are implemented in *Sheet 3* of the Excel file *Chapter18.xls*, together with the chart itself.

The lines plotting the right-hand side of equation 18.8a are generated assuming particular values for P, c, k and i, and also a particular natural growth function describing how timber volume S changes over time. The reader is invited to copy this worksheet, and to study the way in which optimised T varies as p (that is, $P - c$), or k changes, *ceteris paribus*.

18.3.2.1 Comparative static analysis

The results of the previous section have shown that in the infinite-rotation model the optimum rotation depends on:

- the biological growth process of the tree species in the relevant environmental conditions;
- the interest (or discount) rate (i);
- the cost of initial planting or replanting (k);
- the net price of the timber (p), and so its gross price (P) and marginal harvesting cost (c).

Comparative static analysis can be used to make qualitative predictions about how the optimal rotation changes as any of these factors vary. We do this algebraically using equation 18.8b. Derivations of the results are given in Appendix 18.2. Here we just state the results (for convenience, they are tabulated in Table 18.6) and provide some intuition for each of them.

Table 18.6 The infinite-rotation model: comparative static results

Change in:	i	k	$p = P - c$
Effect on optimal rotation length	$dT/di < 0$	$dT/dk > 0$	$dT/dp < 0$

Changes in the interest rate

The result that $dT/di < 0$ means that the interest rate and the optimal rotation period are negatively related. An increase (decrease) in i causes a decrease (increase) in T. Why does this happen? Once planted, there are costs and benefits in leaving a stand unfelled for a little longer. The marginal benefit derives from the marginal revenue product of the additional timber growth. The marginal costs are of two kinds: first, the interest earnings forgone in having capital (the growing timber) tied up a little longer; and second, the interest earnings forgone from not clearing and then selling the bare land at its capital (site) value. If the interest rate increases, the terms of this trade-off change, because the opportunity costs of deferring felling become larger.[12] Foresters respond to this by shortening their forest rotation period.

Changes in planting costs

The result that $dT/dk > 0$ means that a change in planting costs changes the optimal rotation in the same direction. A fall in k, for example, increases the site value of the land, Π. With planting costs lower, the profitability of all future rotations will rise, and so the opportunity costs of *delaying* replanting will rise. The next replanting should take place sooner. The optimal stand age at cutting will fall.

Changes in the net price of timber

The result that $dT/dp < 0$ means that the net price of timber (p) and the optimal rotation length are negatively related. Therefore, an increase in timber prices (P) will decrease the rotation period, and an increase in harvest costs will increase the rotation period. We leave you to deduce the intuition behind this for yourself, in the light of what we have suggested for the two previous cases.

An Excel spreadsheet model (*palc18.xls*) can be used to explore these changes *quantitatively*, for an assumed growth process and particular values of the relevant economic parameters. We recommend that you work through that Excel file, and then experiment further with it. The workbook allows you to reproduce the numbers given in the textbook, to answer the Problems at the end of the chapter, and to see how the comparative static results work out quantitatively.

[12] There is a trap to watch out for here. An increase in discount rates will increase the opportunity cost of each unit of tied-up capital; but at the same time, it will reduce the magnitude of Π, which you will recall is measured in present-value terms. However, inspection of equation 18.8.4 in Appendix 18.2 confirms that the effect of a change in i on T must be negative.

18.3.2.2 Comparing single and infinite rotations: how does a positive site value affect the length of a rotation?

To see the effect of land site values on the optimal rotation interval, compare equation 18.9 (the Hotelling rule taking into consideration positive site values) with equation 18.10, which is the Hotelling rule when site values are zero (and is obtained by setting $\Pi = 0$ in equation 18.9):

$$\frac{p\dfrac{\mathrm{d}S}{\mathrm{d}T}}{pS_T} = i \qquad (18.10)$$

In this case, an optimal rotation interval is one in which the rate of growth of the value of the growing timber is equal to the interest rate on capital alone.

But it is clear from inspection of equation 18.9 that for any given value of i, a positive site value will mean that $(\mathrm{d}S/\mathrm{d}t)/S$ will have to be larger than when the site value is zero if the equality is to be satisfied. This requires a shorter rotation length, in order that the rate of timber growth is larger at the time of felling. Intuitively, the opportunity cost of the land on which the timber is growing requires a compensating increase in the return being earned by the growing timber. With fixed timber prices, this return can only be achieved by harvesting at a point in time at which its biological growth is higher, which in turn requires that trees be felled at a younger age. Moreover, the larger is the site value, the shorter will be the optimal rotation.

It is this which explains why the optimal rotation intervals (for forests that are commercially viable) shown in Table 18.4 are shorter for infinite rotations than for a single rotation. In an infinite-rotation model, land is valuable (because the timber that can be grown on it in the future can yield profits), and the final term in equation 18.9 comes into play.

The reader should note that the way in which bare land is valued by the Faustmann rule – the present value of profits from an infinite sequence of optimal timber rotations – is not the only basis on which one might choose to arrive at land values. Another method would be to value the land at its true opportunity cost basis – that is, the value of the land in its most valuable use other than forestry. In many ways, this is a more satisfactory basis for valuation. This approach can give some insights into forestry location. In remote areas with few alternative land uses, low land prices may permit commercial forest growth even at high altitude where the intrinsic rate of growth of trees is low. In urban areas, by contrast, the high demand for land is likely to make site costs high. Timber production is only profitable if the rate of growth is sufficiently high to offset interest costs on tied-up land capital costs. There may be no species of tree that has a fast enough growth potential to cover such costs. In the same way, timber production may be squeezed out by agriculture where timber growth is slow relative to crop potential (especially where timber prices are low). All of this suggests that one is not likely to find commercial plantations of slow-growing hardwood near urban centres unless there are some additional values that should be brought into the calculus. It is to this matter that we now turn.

18.4 Multiple-use forestry

In addition to the timber values that we have been discussing so far, forests are capable of producing a wide variety of non-timber benefits. These include a variety of 'protective' functions, including soil and water conservation, avalanche control, sand-dune stabilisation, desertification control, coastal protection, and climate control. Non-timber benefits also include food items (fruits, nuts), vegetable products (latex, vegetable ivory), firewood, habitat support for a biologically diverse system of animal and plant populations, wilderness existence values, and a variety of recreational and aesthetic amenities.[13] Where forests do provide one or more of these benefits to

[13] Climate control derives from the role of forests as carbon sinks. FRA 2005 reports estimates that the world's forests (not only plantation forests, note) store 283 gigatonnes (Gt) of carbon in their biomass alone, and that the carbon stored in forest biomass, dead wood, litter and soil together is more than the amount of carbon in the atmosphere. It also reports that, for the world as a whole, carbon stocks in forest biomass decreased by 1.1 Gt of carbon annually over the 5 years to 2005, owing to continued deforestation and forest degradation, partly offset by forest expansion (including planting) and an increase in growing stock per hectare in some regions.

a significant extent, they are called multiple-use forests.[14]

Where plantation forests are managed exclusively for their commercial values, the range and magnitude of these non-timber benefits is likely to be substantially lower than would be the case of equivalent amounts of natural or semi-natural forest. Plantation forests run on purely commercial principles will tend to be planted with fast-growing non-native species, and with little variation in tree species and density of coverage.

Nevertheless, plantation forests can be managed in ways that are capable of substantially increasing the magnitude and variety of non-timber benefits. Efficiency considerations imply that the choices of how a forest should be managed and how frequently it should be felled (if at all) should take account of the multiplicity of forest uses. If the forest owner is able to appropriate compensation for these non-timber benefits, those benefits would be factored into his or her choices and the forest should be managed in a socially efficient way. If these benefits cannot be appropriated by the landowner then, in the absence of government regulation, we would not expect them to be brought into the owner's optimising decisions. Decisions would be privately optimal but socially inefficient. For the moment we will assume that the owner can appropriate the value generated by all the benefits of the forest: both timber and non-timber benefits.

In the rest of this section, we consider just one aspect of forest management (by no means the only one) that can have important implications for the amount of non-timber benefits that a forest is capable of yielding: the rotation period of the forest. Our first task is to work out how the inclusion of these additional benefits into the calculations alters the optimal rotation age of a forest. Once again we imagine beginning at time zero with some bare land. Let NT_t denote the *undiscounted* value of the flow of non-timber benefits t years after the forest is established. The present value of these non-timber value flows over the whole of the first rotation of duration T is

$$\int_{t=0}^{t=T} \mathrm{NT}_t \mathrm{e}^{-it} \mathrm{d}t$$

Now for simplicity denote this integral as N_T, so that we regard the present value of the stream of non-timber values (N) during one rotation as being a function of the rotation interval (T). Adding the present value of the non-timber benefits to the present value of timber benefits, the present value of all forest benefits for the first rotation is

$$\mathrm{PV}_1 = (pS_T - k)\mathrm{e}^{-iT} - k + N_T$$

For a single rotation, the optimal age at which the stand should be felled is that value of T which maximises PV_1. Is the rotation age lengthened or shortened? In this special case (a single rotation only) the answer is unambiguous. Provided that non-timber values are positive, the optimal felling age will be increased. This is true irrespective of whether the non-timber values are constant, rising or falling through time. To see why, note that if these values are always positive, the NPV of non-timber benefits will increase the longer is the rotation. This must increase the age at which it is optimal to fell the forest. Problem 5 at the end of this chapter invites you to use an Excel file, *Non timber.xls*, to explore this matter and verify these conclusions.

Matters are more complicated in the case of an infinite succession of rotations of equal duration. Then the present value of the whole infinite sequence is given by

$$\begin{aligned}\Pi^* = {} & [pS_T\mathrm{e}^{-iT} - k + N_T] \\ & + \mathrm{e}^{-iT}[pS_T\mathrm{e}^{-iT} - k + N_T] \\ & + \mathrm{e}^{-2iT}[pS_T\mathrm{e}^{-iT} - k + N_T] \\ & + \mathrm{e}^{-3iT}[pS_T\mathrm{e}^{-iT} - k + N_T] \\ & + \ldots \end{aligned} \tag{18.11}$$

[14] FRA 2005, for which countries reported on (among other things) the area of forest in which conservation of biological diversity was designated as the primary function (Figure 12), reports that this area has increased by an estimated 96 million hectares since 1990 and now accounts for 11 percent of total forest area. These forests are mainly, but not exclusively, located within protected areas. Conservation of biological diversity was reported as one of the management objectives (primary or secondary) for more than 25% of the forest area. FRA 2005 also estimates that 348 million hectares of forests have a protective function as their primary objective, and that the overall proportion of forests designated for protective functions increased from 8% in 1990 to 9% in 2005. For the use of forests for recreation, tourism, education and conservation of cultural and spiritual sites, Europe – the only continental region where FAO has reliable and comprehensive data – the provision of such 'social' services was reported as the **primary** management objective for 2.4% of total forest area.

which is just a generalisation of equation 18.4 including non-timber benefits. Alternatively, we could interpret equation 18.11 as saying that the present value of all benefits from the rotation (Π^*) is equal to the sum of the present value of timber-only benefits from the rotation (Π) and the present value of non-timber-only benefits from the infinite sequence of rotations.

A forest owner who wishes to maximise the net present value of timber and non-timber benefits will choose a rotation length that maximises this expression. Without going through the derivation (which follows the same steps as before), wealth maximisation requires that the following first-order condition is satisfied:

$$p\frac{\mathrm{d}S}{\mathrm{d}T^*} + N^*_T = ipS_{T^*} + i\Pi^* \qquad (18.12)$$

in which asterisks have been included to emphasise the point that the optimal rotation interval when all benefits are considered (T^*) will in general differ from the interval which is optimal when only timber benefits are included in the function being maximised (T). For the same reason, the optimised present value (and so the land site value) will in general be different from their earlier counterparts, and we will denote these as Π^*.

What effect does the inclusion of non-timber uses of forests have on the optimal rotation length? Inspection of equation 18.12 shows that non-timber benefits affect the optimal rotation in two ways:

- the present value of the flows of non-timber benefits over any one rotation (N^*_T) enters equation 18.12 directly; other things being equal, a positive value for N^*_T implies a reduced value of $\mathrm{d}S/\mathrm{d}T$, which means that the rotation interval is lengthened;
- positive non-timber benefits increase the value of land (from Π to Π^*) and so increase the opportunity cost of maintaining timber on the land; this will tend to reduce the rotation interval.

Which of these two opposing effects dominates depends on the nature of the functions $S(t)$ and $N(t)$. Therefore, for infinite-rotation forests it is not possible to say *a priori* whether the inclusion of non-timber benefits shortens or lengthens rotations. However, some qualitative results can be obtained from equation 18.8(b), which for convenience is given again here:

$$p\frac{\mathrm{d}S_T}{\mathrm{d}T} = ipS_T + i\Pi$$

Recall that Π is called the site value of the land, and is equal to the maximised present value of an endless number of stands of timber that could be grown on the land. The second term on the right-hand side – often called land rent – is thus the interest forgone by not selling the land at its current site value. The first term on the right-hand side constitutes the interest forgone on the value of the growing timber. Adding these two costs together, we arrive at the full opportunity cost of this choice, the marginal cost of deferring harvesting. The left-hand side is the increase in the net value of the timber left growing for an additional period, and so is the marginal benefit of deferring harvesting. An efficient choice equates the values of these marginal costs and benefits.

This equality is represented graphically in Figure 18.5. The inclusion of non-timber values potentially changes the left-hand side of equation 18.8b. If non-timber values are greater in old than in young forests (are rising with stand age) then non-timber values have a positive annual increment; adding these to the timber values will increase the magnitude of the change in overall (timber + non-timber) benefits, shifting the incremental benefits curve upwards. Its intersection with the incremental costs curve will shift to the right, generating a longer optimal rotation. An equivalent, but opposite, argument shows that falling non-timber benefits will shorten the optimal rotation.

Only if the flow of non-timber benefits is constant over the forest cycle will the optimal rotation interval be unaffected. Hence it is variation over the cycle in non-timber benefits, rather than their existence as such, that causes the rotation age to change.

It is often assumed that NT (the annual magnitude of undiscounted non-timber benefits) increases with the age of the forest. While this may happen, it need not always be the case. Studies by Calish *et al.* (1978) and Bowes and Krutilla (1989) suggest that some kinds of non-timber values rise strongly with forest age (for example, the aesthetic benefits of forests), others decline (including water values) and

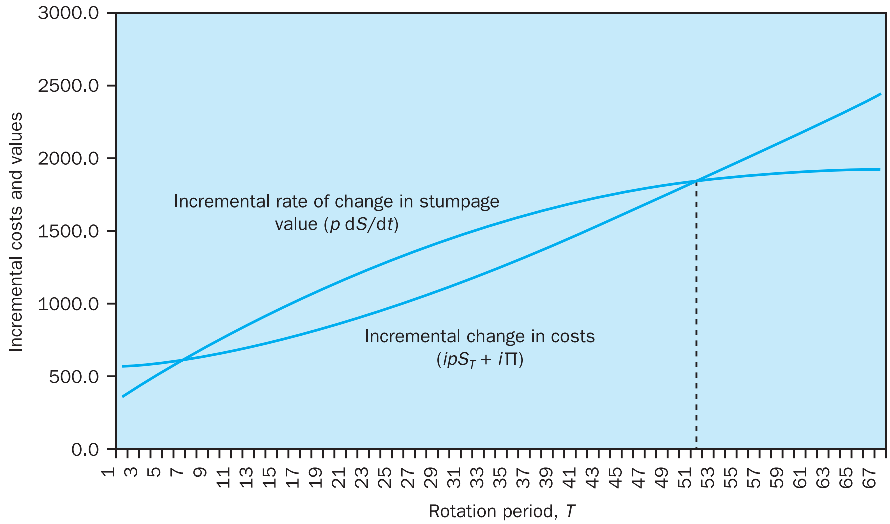


Figure 18.5 Incremental change in value and costs with rotation stand age

yet others have no simple relationship with forest age. There is also reason to believe that total forest benefits are maximised when forests are heterogeneous (with individual forests being specialised for specific purposes) rather than being managed in a uniform way (see Swallow and Wear, 1993; Vincent and Blinkley, 1993). All that can be said in general is that it is most unlikely that total non-timber benefits will be independent of the age of forests, and so the inclusion of these benefits into rotation calculations will make some difference.

Note also that in extreme cases the magnitude and timing of non-timber benefits may be so significant as to result in no felling being justified. Where this occurs, we have an example of what is called 'dominant-use' forestry. It suggests that the woodland in question should be put aside from any further commercial forest use, perhaps being maintained as a national park or the like.

As a matter of interest at a time when reducing the growth of carbon dioxide atmospheric concentration is so central to international environmental policy, we note that CO_2 sequestration varies with the growth rate and so favours shorter rotations, given that growth slows right down with old age. This is not good news for mature natural forests; if CO_2 sequestration were our sole concern, then the best thing would be to chop down mature forests and plant new ones. There are some qualifications to this kind of reasoning; for example, we might need to ensure that the felled mature timber would be locked up in new built houses or furniture. But this is suggestive of a case where there could be a trade-off between climate change mitigation and biodiversity conservation.

18.5 Socially and privately optimal multiple-use plantation forestry

Our discussions of multiple-use forestry have assumed that the forest owner either directly receives all the forest benefits or is able to appropriate the values of these benefits (presumably through market prices). In that case, what is privately optimal will also be what is socially optimal (provided, of course, that there is no divergence between social and private consumption discount rates). But it is most implausible that forest owners can appropriate all forest benefits. Many of these are public goods; even if exclusion could be enforced and markets brought into existence, market prices would undervalue the marginal social benefits of those public goods. In many circumstances, exclusion will not be possible and open-access conditions will prevail.

Where there is a divergence between private and social benefits, the analysis of multiple-use forestry we have just been through is best viewed as providing information about the socially optimal rotation length. In the absence of efficient bargaining (see Chapter 4), to achieve such outcomes would involve public intervention. This might consist of public ownership and management, regulation of private behaviour, or the use of fiscal incentives to bring social and private objectives into line.

Whether public or private ownership of forest land is better for socially efficient forest management is not clear. On the one hand, one might suppose that the incentives facing owners and their agents in privately owned forests would not be well aligned with including non-timber benefits in forest management choices. Some non-timber benefits (such as forest recreation) are private goods that are capable of being marketed; but the public good property of several other non-timber benefits makes this option unviable.

The absence of a profit-maximisation motive in public ownership does not seem to present such conflicts of interest. However, public ownership of forest assets is not without its problems. Often public ownership is associated with *de facto* open-access conditions, and governments may well be inclined to let short-term economic and political gains from consumptive uses of forests determine policy choices. Thus a move away from public ownership towards private ownership (with forested land being partitioned into private 'estates' that are not excessively sized) and in which taxation or subsidy schemes are offered to internalise the non-timber benefit externalities may well be the best way forward in practice in many countries.

FRA 2005 notes that 84% of the world's forests are publicly owned, but that private ownership is increasing. It reports that 'trends seen over the past 20 years towards community empowerment, decentralized decision-making and increased involvement of the private sector in forest management are reflected in some regions in changes in forest ownership and tenure. . . . .Differences among regions are considerable. North and Central America, Europe (apart from the Russian Federation), South America and Oceania have a higher proportion of private ownership than other regions.'

In summary, the fact that forest land often satisfies multiple uses suggests that there are likely to be efficiency gains available where government integrates environmental policy objectives with forestry objectives, whether that be through active and responsible public ownership or through private ownership with appropriate incentive mechanisms in place.

18.6 Natural forests and deforestation

A series of recent studies, including FAO (1995), FAO (2001), FRA 2005, and various editions of *World Resources* (by the World Resources Institute), paint a vivid picture of the pattern and extent of natural forest loss and conversion (deforestation). The extent of human impact on the natural environment can be gauged by noting that by 2000 approximately 40% of the earth's land area had been converted to cropland and permanent pasture. Most of this has been at the expense of forest and grassland.

Until the second half of the twentieth century, deforestation largely affected temperate regions. In several of these regions, the conversion of temperate forests has been effectively completed. North Africa and the Middle East now have less than 1% of land area covered by natural forest. It is estimated that only 40% of Europe's original forestland remains, and most of what currently exists is managed secondary forest or plantations. The two remaining huge tracts of primary temperate forest – in Canada and Russia – are now being actively harvested, although rates of conversion are relatively slow. Russia's boreal (coniferous) forests are now more endangered by degradation of quality than by quantitative change. The picture is not entirely bleak, however. China has recently undertaken a huge reforestation programme, and the total Russian forest area is currently increasing. And in developed countries, management practices in secondary and plantation forests are becoming more environmentally benign, partly as a result of changing public opinion and political pressure.

Not surprisingly, the extent of deforestation tends to be highest in those parts of the world which have the greatest forest coverage. With the exceptions of temperate forests in China, Russia and North America, it is tropical forests that are the most extensive. And it is deforestation of primary or natural tropical forests that is now the most acute problem facing forest resources. However, there is a large spread of estimates about recent and current rates of loss of tropical rainforest, and about what proportion of original primary tropical forest has been lost. FAO (2001) reports that in the 30 years from 1960 to 1990 one-fifth of all natural tropical forest cover was lost, and that the rate of deforestation increased steadily during that period. However, it also tentatively suggests that this rate may have slightly slowed in the final decade of the twentieth century. To give a flavour of the disparity of results, Raintree, a rainforest conservation organisation, contends that 'Rainforests once covered 14% of the earth's land surface; now they cover a mere 6% . . .'.[15] Analysis of some recent satellite has revealed that deforestation of the Amazon rainforest is twice as fast as scientists previously estimated.[16] In contrast, Bjørn Lomborg claims that global forest cover has remained approximately stable since the middle of the twentieth century, and others have claimed that for every acre of rain forest cut down each year, more than 50 acres of new forest are growing in the tropics.[17]

Box 18.3 Tropical deforestation

Tropical deforestation has many adverse consequences. As far as the countries in which the forests are located are concerned, valuable timber assets are irretrievably lost, and the loss of tree cover (particularly when it is followed by intensive agriculture or farming) can precipitate severe losses of soil fertility. Indigenous people may lose their homelands (and their distinctive cultures), water systems may be disrupted, resulting in increased likelihood of extreme hydrological conditions (more droughts and more floods, for example), and local climates may be subtly altered. Perhaps most pernicious are the losses in potential future incomes which deforestation may lead to. Tropical forests are immense stores of biological diversity and genetic material, and quasi-option values (see Chapters 12 and 13) are forfeited as this diversity is reduced. With the loss of animal and plant species and the gestalt of a primary tropical forest will go recreational amenities and future tourism potential.

All of this reinforces a point made earlier: tropical forests are multiple-service resources *par excellence*. Many of these forest services benefit the world as a whole rather than just local inhabitants. Of particular importance here are lost stores of diverse genetic material, the climate control mechanisms that are part of tropical forest systems, and the emission of greenhouse gases when forests are cleared (see Chapter 9 for further details).

Given these adverse consequences, why are tropical forests being lost? There appears to be no single, predominant cause. As with earlier discussions of biodiversity loss, it is convenient to distinguish between proximate (or immediate) causes and fundamental causes. Economists tend to focus on the latter. Important in this latter category – especially for tropical forests – is the absence of clearly defined and enforceable property rights. The lack of access restrictions must at least partially explain the finding of a study that less than 0.1% of tropical logging is being done on a sustainable yield basis (WR, 1996).

Many commentators give a large role to population pressure, especially when significant numbers of people in burgeoning populations have no land entitlement or are living close to the margin of poverty. However, it is now being realised that too much weight has been attributed to this cause, and that emphasis has been given to it in part at least because most models of deforestation have been constructed to be population-driven (see FAO, 2001). This reflects a point well worth remembering about economic modelling: what you get out (here the

[15] This quote is taken from its 'Rainforest Facts' web page at http://www.rain-tree.com/facts.htm (accessed on 19 January 2010).

[16] Sources: Jha, Alok. 'Amazon rainforest vanishing at twice rate of previous estimates', *The Guardian*, 21 October 2005; 'Satellite images reveal Amazon forest shrinking faster' at http://www.csmonitor.com/2005/1021/p04s01-sten.html, csmonitor.com.

[17] Lomborg references: http://www.econlib.org/library/Enc/EnvironmentalQuality.html, and Lomborg (2001). Other reference: 'New Jungles Prompt a Debate on Rain Forests', *The New York Times*, 30 January, 2009.

Box 18.3 ***continued***

conclusions) depends very much on what you put in (here, modelling structures and assumptions).

Nevertheless, it is not difficult to understand why many governments, faced with growing populations, mounting debt and growing problems in funding public expenditure, will tend to regard tropical forests as capital assets that can be quickly turned into revenues. Moreover, cleared forestland (a state greatly facilitated by road building projects that open up forest land) can also provide large additional sources of land for agriculture and ranching, each of which may offer far greater financial returns than are obtainable from natural forests.

This suggests that the conversion of forestland to other uses (principally agriculture) may well be optimal from the point of view of those who make land-use choices in tropical countries. Of course, it may be the case that the incentive structures are perverse, as a result of widespread market failure. Tropical deforestation is *not* simply the result of ignorance, short-sightedness, or commercial pressure from organised business (although any of these may have some bearing on the matter). It is the result of the patterns of incentives that exist. This way of thinking is important because it suggests ways of changing behaviour, based on altering those incentive structures.

Several writers have developed models of tropical forest conversion arising from optimising rational behaviour. Hartwick (1992) suggests that the use of any single piece of land will be determined by the relative magnitudes of B^F, the net benefits of the land in forestry (which includes both timber and non-timber values) and B^A, the net benefits of the land in agriculture. At the level of the whole economy, there will be many individual natural forest stands, and we can envisage deforestation as a gradual process by which an increasing proportion of these stands is converted to agriculture over time. The socially efficient rate of conversion at any point in time is that at which these benefits are equalised at the margin. That is $MB^F = MB^A$. One might expect MB^F to rise as the remaining area of tropical forest becomes ever smaller. This would tend to slow down forest conversion. However, this effect may be offset by a rise in MB^A which could arise because of population increases or higher incomes. It is not inconceivable that the outcome of this process would be one in which all forestland is converted. That likelihood is increased if MB^F only includes timber benefits, but excludes the non-timber, or environmental, benefits. For the reasons we gave in the text, there are good reasons to believe that the non-timber benefits will be excluded from the actual ‘optimising’ exercise.

Barbier and Burgess (1997) develop Hartwick’s ideas a little further. Their optimising model specifies a demand-and-supply function for forestland conversion to agriculture. At any point in time, the supply and demand for forestland conversion, taking account of both timber and non-timber benefits, can be represented by the functions labelled S_t^* and D_t^* in Figure 18.6. The price shown on the vertical axis is the opportunity cost of land converted to agriculture: that is, forgone timber and non-timber benefits. The demand function is of the form:

$$D = D(P, Y, \mathrm{POP}, Q)$$

where Y is income, POP is the level of population and Q is an index of agricultural yields. Barbier and Burgess expect that $\mathrm{d}D/\mathrm{dPOP}$ is positive, and so population increases will shift the demand curve rightwards, thus increasing deforestation.

If, however, forest owners are unable to appropriate non-timber benefits, the supply curve will shift to the right relative to that shown

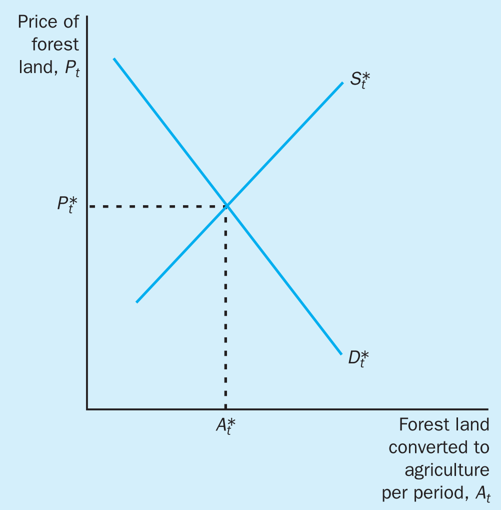


Figure 18.6 The optimal rate of conversion of forested land at time t

Box 18.3 ***continued***

in the diagram (which supposes that both timber and non-timber benefits are appropriable by forest owners). Clearly this would also increase the rate of deforestation (by depressing the price of forestland).

We mentioned in the text that the non-timber benefits of tropical forests are received by people throughout the world, not just in the forest vicinities. The benefits are global environmental goods. An interesting attempt to estimate the size of these benefits has recently been made. Kramer and Mercer (1997) used a contingent valuation approach (see Chapter 12) to estimate the size of the one-off monetary payment that US residents would be willing to pay to conserve 5% of tropical forests. Kramer and Mercer's survey responses gave an average value per household of between $21 and $31. Aggregated over the US population, this is equivalent to a total single payment of between $1.9 billion and $2.8 billion.

It was noted earlier that natural (or primary) forests warrant a very different form of treatment from that used in investigating plantation forestry. Natural forest conversion is something akin to the mining of a resource. These forests represent massive and valuable assets, with a corresponding huge real income potential. While it is conceivable that a forest owner might choose to extract the sustainable income that these assets can deliver, that is clearly not the only possibility. In many parts of the world, as we noted earlier, these assets were converted into income a long time ago. In others, the assets were left almost entirely unexploited until the period after the Second World War. What appears to be happening now is that remaining forest assets are being converted into current income at rates far exceeding sustainable levels.

Where a natural forest is held under private property, and the owner can exclude others from using (or extracting) the forest resources, the management of the resource can be analysed using a similar approach to that covered in Chapter 15 (on non-renewable resources). The basic point is that the owner will devise an extraction programme that maximises the present value of the forest. Whether this results in the forest being felled or maintained in its natural form depends on the composition of the benefits or services the forest yields, and from which of these services the owner can appropriate profits.

Where private ownership exists, the value of the forest as a source of timber is likely to predominate in the owner's management plans even where the forest provides a multiplicity of socially valuable services. This is because the market mechanism does not provide an incentive structure which reflects the relative benefits of the various uses of the forest. Timber revenues are easily appropriated, but most of the other social benefits of forestry are external to the owner. The signals given to owners by the relative returns to the various forest services lead to a socially inefficient allocation of resources, as we explained in Chapter 4 in discussing the consequences of externalities and public goods. These mechanisms go a long way to explain why the rate of conversion of natural forests is so high, why forestland is often inefficiently converted to other land uses, and why the incentives to replant after clearing are sometimes too low to generate reforestation or to ensure its success.

Our arguments have been premised on the assumption that forestland is privately owned and its use correspondingly controlled. But this analysis is of little relevance in circumstances where forests are not privately owned or where access cannot be controlled. There are two main issues here: the first is the consequence of open-access conditions, and the second is the temptation to 'mine' forests for quick returns.

Many areas of natural forest are *de facto* open-access resources. There is no need to repeat the analysis in Chapter 17 of the consequences of open access for renewable resource exploitation. However, in some ways, the consequences will be more serious in this instance. We argued that open-access fisheries have a built-in defence against stocks being driven to zero: as fish numbers decline to low levels, marginal harvesting costs rise sharply. It usually becomes uneconomic to harvest fish to the point where stock levels have reached critical minimum levels. This does not apply in the case of woodland, however. Trees are not mobile and harvesting costs tend to be affected very little by the stock size. So as long as timber values are high (or the return from other uses of the land is sufficiently attractive), there is no in-built mechanism stopping stock declining to zero. Open access also implies that few individuals are willing to incur the large capital costs in restocking felled timber, particularly when returns are so far into the future.

The second issue we raised above was the temptation of governments and individuals granted tenure of land to convert natural timber assets into current income, or to switch land from forestry to another use which offers quicker and more easily appropriated returns. In some instances clearing land is a way of asserting property rights. There is, of course, nothing new about this. It has been happening throughout history, and goes a long way to explaining the loss of natural forest cover in Europe, North Africa and the Middle East. The process is now most acute in tropical forests.

18.7 Government and forest resources

Given the likelihood of forest resources being inefficiently allocated and unsustainably exploited, there are strong reasons why government might choose to intervene in this area. For purely single-use plantation forestry, there is little role for government to play other than guaranteeing property rights so that incentives to manage timber over long time horizons are protected.

Where forestry serves, or could serve, multiple uses, there are many important questions on which government might attempt to exert influence. Important questions include what is to be planted (for example, deciduous or coniferous, or some mixture) and where forest or woodland is to located (so that it is convenient for recreational and other non-timber purposes). The issue of optimal rotation length is also something that government might take an interest in, perhaps using fiscal measures to induce managers to change rotation intervals. It is straightforward to see how this can be done. Well-designed taxes or subsidies can be thought of as changing the net price of timber (by changing either the gross price, P, or the marginal harvest cost, c). In principle, any desired rotation length can be obtained by an appropriate manipulation of the after-tax net price. The suggestions for further reading at the end of this chapter point to some analyses of the relative advantages of alternative forms of taxation and subsidy instruments for achieving forest policy objectives.

Where non-timber values are large and their incidence is greatest in mature forests, no felling may be justified. Government might seek such an outcome through fiscal incentives, but it may prefer to do so through public ownership. The most important role for government, though, concerns its policy towards natural forestland. It is by no means clear that public ownership *per se* has any real advantages over private ownership in this case. What matters here is how the assets are managed, and what incentive structures exist.

Finally, we need to give some attention to international issues here. Many of the non-timber values of forest resources are derived by people living not only outside the forest area but also in other countries. Many of the externalities associated with tropical deforestation, for example, cross national boundaries. This implies limits to how much individual national governments can do to promote efficient or sustainable forest use. Internationally concerted action is a prerequisite of efficient or sustainable outcomes. We discussed these issues – including internationally organised tax or subsidy instruments, debt-for-nature swap arrangements and international conservation funds – in Chapter 9.

Summary

- If all markets exist, all the conditions described in Chapter 4 for the efficient allocation of resources are satisfied throughout the economy, and if the interest rate used by private foresters is identical to the social consumption discount rate, privately optimal choices in forestry will be socially efficient, and, given appropriate distributions of initial endowments of property rights, could be socially optimal too.
- These conditions are not likely to be satisfied. Apart from the fact that the 'rest of the economy' is unlikely to satisfy all the necessary efficiency conditions, there are particular aspects of forestry that imply a high likelihood of private decisions not being socially efficient. What are these aspects?
  1. Where forests are privately owned, externalities tend to drive a wedge between privately and socially efficient incentive structures whenever forests serve multiple uses. Forests are multi-functional, providing a wide variety of economic and other benefits. Private foresters are unlikely to incorporate all these benefits into their private net benefit calculations, as they often have very weak or no financial incentives to do so. Non-timber benefits may be very substantial. Where plantation forests are being managed, the presence of these benefits is likely to cause the length of socially optimal rotations and of choices about what to plant and where to plant to diverge from what is privately optimal.
  2. In the case of natural forests, it will also be difficult for whoever has responsibility for landuse decisions to extract appropriate monetary values for these non-timber benefits, particularly when the benefits are received by citizens of other countries. These problems are particularly acute in the case of tropical forests and other open-access woodlands.
- Governments might attempt to internalise externalities by fiscal measures or by the regulation of land use. Alternatively, public ownership of forestland may be used as a vehicle for promoting socially efficient forest management.

Further reading

Excellent reviews of the state of forest resources in the world economy, and experiences with various management regimes, are contained in *World Resources*, published every two years. See, in particular, the sections in WR (1994) and WR (1996). This source also contains an excellent survey concerning trends in biodiversity. Various editions of the *United Nations Environment Programme, Environmental Data Report* also provide good empirical accounts. Extensive references on biodiversity were given in Chapter 17.

A more extensive account of forestry economics (at about the same level as this text), examining the effects of various tax and subsidy schemes, is to be found in Hartwick and Olewiler (1998), chapter 10. Other excellent surveys of the economics of forestry can be found in Anderson (1991), Pearse (1990), Berck (1979) and Johansson and Löfgren (1985). Montgomery and Adams (1995) contains a good account of optimal management, but at a relatively advanced level.

Bowes and Krutilla (1985, 1989) are standard references for multiple-use forestry. Hartman (1976) is an early work in the area, which is also examined in Calish *et al*. (1978), Swallow *et al*. (1990), Swallow and Wear (1993), Pearce (1994) and Vincent and Blinkley (1993).

The value of forests for recreation is analysed by Clawson and Knetsch (1966), Benson and Willis (1991) and Cobbing and Slee (1993), although you should note that these references are primarily concerned with the techniques of valuation of non-marketed goods that we discuss in Chapter 12. Browder (1988) examines the conversion of forest-land in Latin America. The state of tropical and other natural-forest resources, with an emphasis on sustainability and policy, is discussed in Sandler (1993), Barbier and Burgess (1997), Vincent (1992) and Repetto and Gillis (1988). For the effects of acid rain on forests, see CEC (1983) and Office of Technology Assessment (1984).

Tahvonen and Salo (1999) present a synthesis of the Fisher two-period consumption-saving model with the Faustmann model, thereby allowing owner preferences to shape forest management choices.

Discussion questions

1. Is it reasonable for individuals living in Western Europe today to advise others to conserve tropical forests given that the countries in which they live effectively completed the felling of their natural forests centuries ago?
2. Discuss the implications for the harvest rate and possible exhaustion of a renewable resource under circumstances where access to the resource is open, and property rights are not well defined.
3. Discuss the contention that it is more appropriate to regard natural forests as non-renewable than as renewable resources.
4. In what circumstances, and on what criterion, can the conversion of tropical forestry into agricultural land be justified?
5. How will the optimal rotation interval be affected by extensive tree damage arising from atmospheric pollution?

Problems

1. Using a spreadsheet program, calculate the volume of timber each year after planting for a period of up to 130 years for a single unfelled stand of timber for which the age–volume relationship is given by $S = 50t + 2t^2 - 0.02t^3$ (where S and t are defined as in the text of this chapter). Is it meaningful to use this equation to generate stock figures up to this stand age?

   Also calculate:
   (a) The year after planting at which the amount of biological growth, $G(S)$, is maximised.
   (b) The present-value-maximising age for clear felling (assuming the stand is not to be replanted) for the costs and prices used in Table 18.4 and a discount rate of 5%.

   (We suggest that you attempt to construct your own spreadsheet program to answer this question. If you find that this is not possible, you can obtain the answers by adapting *Sheet 4* in *Chapter18.xls*.)
2. Demonstrate that a tax imposed on each unit of timber felled will increase the optimal period of any rotation (that is, the age of trees at harvesting) in an infinite-rotation model of forestry. What effect would there be on the optimal rotation length if the expected demand for timber were to rise?
3. How would the optimal rotation interval be changed as a result of
   (a) an increase in planting costs;
   (b) an increase in harvesting costs;
   (c) an increase in the gross price of timber;
   (d) an increase in the discount rate;
   (e) an increase in the productivity of agricultural land?

4. The following three exercises require that you use the Excel file *palc18.xls*.
   (a) Calculate the optimal rotation lengths for a single-rotation forest for the interest rates 1, 2, 4, 5 and 6%. These should match those shown in Table 18.5.
   (b) Calculate the interest rate above which the PV of the forest becomes negative for *any* rotation length in a single rotation forest. Do the same for an infinite-rotation forest.
   (c) Identify what happens to the gap between the optimal rotation lengths in single- and infinite-rotation models as the interest rate becomes increasingly large (beginning from 0%). Explain the convergence that you should observe. What happens to the PV of the forest at this convergence?

5. The Excel workbook *Non Timber.xls* (see Companion Website) models the consequences of including non-timber values in a single-rotation forest model. The first sheet – *Parameter values* – defines various parameter values, and gives three alternative sets of non-timber present values. Results of the computations are shown in *Sheet 1*. Examine how the inclusion of non-timber benefits alters the optimal stand age at which felling takes place. Does the change vary from one set of non-timber values to another? Do your conclusions differ between the cases where the discount rate is 2% and 4%?

6. Equation 18.8 in the text was given in two forms. The first was given by

$$\frac{p\dfrac{\mathrm{d}S}{\mathrm{d}T}}{pS_T - k} = \frac{i}{1 - \mathrm{e}^{-iT}} \tag{18.8a}$$

and the second was given by:

$$p\frac{\mathrm{d}S_T}{\mathrm{d}T} = ipS_T + i\Pi \tag{18.8b}$$

Find the equivalent first-order conditions for optimal extraction of oil (in Chapter 14) and fish (in Chapter 17). Compare 18.8 (a or b) with those two other results, and try to show what is similar and what is different between them. Note that for fish, the site value will be zero. We shall leave you to ponder whether that is also true for oil.

CHAPTER 19 Accounting for the environment

There is a dangerous asymmetry today in the way we measure, and hence, the way we think about, the value of natural resources. Man-made assets – buildings and equipment, for example – are valued as productive capital and are written off against the value of production as they depreciate. This practice recognizes that a consumption level maintained by drawing down the stock of capital exceeds the sustainable level of income. Natural resource assets are not so valued, and their loss entails no debit charge against current income that would account for the decrease in potential future production. A country could exhaust its mineral resources, cut down its forests, erode its soils, pollute its aquifers, and hunt its wildlife to extinction, but measured income would not be affected as these assets disappeared.

Repetto *et al.* (1989), p. 4

Statisticians are trying to adjust measures of national wealth for pollution and depleted resources. This turns out to be all but impossible. ***The Economist*, 18 April 1998**

Learning objectives

In this chapter you will

- find out about the steps that many countries are taking to use environmental indicators to report the state of the environment
- learn about what economic theory says about defining national income so that what gets measured is sustainable income
- have explained proposals made by national income statisticians with the aim of having the published national income accounts report on resource depletion and environmental degradation
- consider the idea of genuine saving as a sustainability indicator
- encounter the difficulties that arise when trying to measure genuine saving
- learn about some suggested alternatives to national income as a measure of economic performance and sustainable development indicator

Introduction

The development of the national income accounting conventions (mainly in the 1940s and 1950s) took place in a period in which there was less concern about the impact of economic development on the environment. The conceptual basis and scope of the national accounts were governed by definitions of income and wealth which did not make any allowance for the depletion of natural resources or the costs of environmental damage such as pollution.

It is now widely appreciated that production and consumption activities have environmental effects which impose considerable costs, some of which will be borne by future generations. There is a perceived requirement for information that will permit economic activity to be so managed that it is sustainable. This chapter is about emerging responses to that perception. The response that has most engaged economists is directed at modifications to national income accounting conventions such that what would get measured is sustainable income, which, as indicated in the quotation which heads this chapter, is not what currently gets measured. This is one area where there is a high level of agreement between environmental economists and environmental activists, most of whom also want to see changes to the national income accounting conventions. Many economists and environmentalists argue that such changes are essential for the pursuit of sustainability.

Criticism of current accounting conventions centres on three main issues: the absence of any allowance for the depletion of natural resources, the absence of any adjustment for degradation of environmental amenity, and the fact that activity to offset environmental damage is counted as part of income. The work done by economists on ‘environmental accounting’ – also sometimes referred to as ‘natural resource accounting’ or ‘green accounting’ – falls into two distinct, but related, parts. First, as discussed in the second part of this chapter, theoretical economists have used abstract models to consider how income should properly be measured, given the interdependence of the economy and the environment. Second, as discussed in the third part of this chapter, national income statisticians have developed proposals for the modification of the existing accounting conventions.

Closely related to the idea of properly measuring income is that of properly measuring wealth. It has been proposed that the change in a comprehensive measure of wealth, known as ‘genuine saving’ or ‘genuine investment’, can serve as a sustainability indicator. The comprehensive measure of wealth would include environmental assets as well as economic assets. In the terminology introduced in Chapter 2 it would, that is, include natural as well as reproducible capital, together with human capital and social capital. We look at this idea, and some attempts at implementation, in the fourth part of the chapter.

The fifth part reviews some ideas about providing measures of economic performance in relation to the goals of sustainability and sustainable development that do not take income or wealth as their starting point. The final section of the chapter offers some observations on where we think we have got to in terms of the contribution that environmental accounting, and economic analysis more generally, is making, and can make, to the pursuit of sustainable development.

Not everybody who sees the need for information about the natural environment is concerned about sustainable development. There is a demand for biophysical data concerning the state of the environment which exists independently of the demand for measures of economic performance. In any case, such data are a logical prerequisite to monetary data which can be used in economic accounts. Hence, in the first section of the chapter we take a brief look at environmental indicators.

19.1 Environmental indicators and state of the environment reporting

While the terminology is not universal, ‘environmental indicators’ typically refers to biological and/or physical data concerning the natural environment organised around environmental issues, whereas ‘environmental accounting’ typically refers to monetary data, or to biophysical data organised around economic categories. The term ‘environmental statistics’ is sometimes used to refer to biological and/or physical data. Compilations of environmental indicators are often called ‘state of the environment reports’. Most developed nations now produce state of the environment reports.[1] In this section we look briefly at state of the environment reporting in two countries, so as to indicate the scope and nature of this kind of activity. The amount of information that such reports contain is large, and for a proper appreciation of state of the environment reporting the reader is urged to go to one of the sources cited here or in the Further Reading section at the end of the chapter.

[1] In 2007 UNSTAT did a survey of what UN members were doing about ‘environmental statistics’ and ‘environment-economy accounting’: it appears that by ‘statistics’ UNSTAT means what we call ‘indicators’ here. The results are reported at http://unstats.un.org/unsd/statcom/doc07/Analysis_SC.pdf. Of 192 UN members, 88 said they were running environmental statistics programmes, and 49 said they were running environment-economy accounting programmes, of which 29 were developed countries.

19.1.1 Environmental reporting for the USA

In the USA, at the federal level, the Environmental Protection Agency (EPA) is responsible for the protection of human health and the environment. In 2001 the EPA initiated a project to put together a comprehensive set of environmental indicators relevant to its mission. A draft report was issued in 2003, and, following stakeholder feedback and revision, the EPA published, in 2008, the first comprehensive state of the environment report for the USA, with the title *EPA's 2008 Report on the Environment*, hereafter ROE. The report (US Environmental Protection Agency, 2008) can be downloaded from http://www.epa.gov/roe, as can a shorter summary version, *EPA's 2008 Report on the Environment: Highlights of National Trends.* This website is also the point of entry to the ROE Dynamic website, which is to provide data updates as they become available.

Box 1.1 in ROE gives the EPA's definition of an environmental indicator and the criteria that were used to select indicators for inclusion in the ROE. The definition is:

> . . . a numerical value derived from actual measurements of a stressor, state or ambient condition, exposure, or human health or ecological condition over a specified geographic domain, whose trends over time represent or draw attention to underlying trends in the condition of the environment

In summary form the criteria are:

- The indicator is useful and contributes to the answer to an ROE question.
- The indicator is objective.
- The indicator is transparent and reproducible.
- The underlying data are properly collected.
- Data are available over time and are up to date.
- The data are representative, and comparable across time and space.

In the ROE, indicators are reported on in five chapters. Within each chapter, several issues are identified, and corresponding to each issue are one or more of the 23 questions referred to in the first criterion above. These questions are those that the EPA believes should be answered for adequate information about environmental trends. For 22 questions, the ROE provides data on one or more relevant indicators. The ROE notes (pp. 1–3) that questions that '*should* be answered' are 'not necessarily questions that EPA *can* fully answer at present based on the indicators that meet the ROE definition and criteria'. Table 19.1 lists the ROE chapter titles, the 23 questions, and for each question the number of indicators used to answer it. For one question there is no indicator.

In relation to the ambitions of some contributions to the economics literature on environmental accounting and sustainability indicators, to be discussed in following sections, it is worth taking note of some of the comments and assessments from the final chapter of the ROE. It should first be noted that there is no attempt in the ROE to aggregate across indicators to come up with a 'bottom-line' single indicator in relation to a question such as 'Is the state of the US environment sustainable?' or 'Are Americans acting sustainably in regard to their natural environment?'.

The ROE reports (p. 7-2) that:

> There currently are no 'meta-indicators' that can provide an integrated, comprehensive measure of trends in human health or the environment to answer any of the ROE questions. Instead, the available indicators provide in-depth coverage of particular aspects of the environment or health that are relevant to answer the questions.

The EPA cannot at this time, that is, provide a bottom-line answer to any of the questions listed in Table 19.1. It is, it reports, working on improving existing indicators and developing new ones.

The ROE notes the existence of 'emerging issues' which are 'issues whose potential to affect human health and the environment is not well understood'. In such cases, 'the current state of scientific understanding makes it unclear whether indicators are needed, and if so, how they should be constructed and tracked'. In this context they cite new technologies and

> Issues for which the inherent complexity of the interactions between pollutants, environmental media, and ecological systems makes it unclear what should be measured

for which they give as examples climate change and biodiversity loss.

Table 19.1 Coverage of the 2008 ROE for the USA

Chapter	Questions	Indicators
Air	What are the trends in outdoor air quality and their effects on human health and the environment?	23
	What are the trends in greenhouse gas emissions and concentrations?	2
	What are the trends in indoor air quality and their effects on human health?	2
Water	What are the trends in the extent and condition of fresh surface waters and their effects on human health and the environment?	8
	What are the trends in the extent and condition of groundwater and their effects on human health and the environment?	1
	What are the trends in the extent and condition of wetlands and their effect on human health and the environment?	1
	What are the trends in the extent and condition of coastal waters and their effect on human health and the environment?	7
	What are the trends in the quality of drinking water and their effects on human health?	1
	What are the trends in the condition of recreational waters and their effects on human health and the environment?	0
	What are the trends in the condition of consumable fish and their effects on human health?	2
Land	What are the trends in land cover and their effects on human health and the environment?	3
	What are the trends in land use and their effects on human health and the environment?	2
	What are the trends in wastes and their effects on human health and the environment?	2
	What are the trends in chemicals used on the land and their effects on human health and the environment?	4
	What are the trends in contaminated land and their effects on human health and the environment?	2
Human Exposure and Health	What are the trends in human exposure to environmental contaminants, including across population subgroups and regions?	7
	What are the trends in health status in the United States?	3
	What are the trends in human disease and conditions for which environmental contaminants may be a risk factor, including across population subgroups and geographic regions?	9
Ecological Condition	What are the trends in the extent and distribution of the nation's ecological systems?	9
	What are the trends in the diversity and biological balance of the nation's ecological systems?	6
	What are the trends in the ecological processes that sustain the nation's ecological systems?	1
	What are the trends in the critical physical and chemical attributes of the nation's ecological systems?	10
	What are the trends in biomarkers of exposure to common contaminants in plants and animals?	3

Source: US Environmental Protection Agency, (2008), http://www.epa.gov/roe (accessed February 2009)

19.1.2 Environmental indicators for the UK

In the UK the government department responsible for the environment is the Department for Environment, Food and Rural Affairs (DEFRA). It maintains an 'e-Digest of Environmental Statistics' at http://www.defra.gov.uk/environment/statistics, from where one can access data on a variety of environmental topics, and download copies of some of the principal publications. These include what is, in effect, an annual state of the environment report for the UK, the most recent of which is *The environment in your pocket 2008* (DEFRA, 2008a). DEFRA has been producing these reports since 1996, and the content and organisation has varied somewhat over the years.

In the 2008 report there are data on 56 indicators, organised in terms of 11 themes, as shown in Table 19.2 here. As with the ROE for the USA, there is no attempt to aggregate over indicators to produce summary assessments for the environment as a whole, or for the themes treated separately. Looking at Tables 19.1 and 19.2 one can see, despite the different forms of presentation here, that there is, as would be expected, a considerable degree of commonality in coverage. There are, however, some notable differences. The UK report is not directly concerned with human health. It is not exclusively concerned

Table 19.2 UK environmental indicator list 2008

Theme	Indicator
Climate change	Average surface temperature (Global and Central England) 1772–2007 Rainfall and temperature (England and Wales) 1845–2007 Sea level rise at selected sites (UK) 1850–2006 EU emissions of greenhouse gases 1990–2012 Emissions of greenhouse gases (UK) 1990–2007 Carbon dioxide emissions associated with UK consumption 1992–2004 Carbon dioxide emissions by end user (UK) 1990–2006 Greenhouse gas emissions from food chain (UK) 2006 Carbon dioxide emissions from local authority areas (UK) 2006 Methane emissions by source (UK) 1990–2006 Nitrous oxide emissions by source (UK) 1990–2006 Fuel used for electricity generation (UK) 1990–2007 Electricity generated by renewable sources (UK) 1996–2007 Energy consumption per household by end user (UK) 1990–2006 Private car CO_2 emissions and car-kilometres and household spending (UK/GB) 1990–2006 Household car availability (GB) 1989/91–2007
Contextual	Population estimates and projection by age group (UK) 1981–2036 Household estimates and projections by household type (England) 2004–2029
Public attitudes and behaviours	Attitudes to the environment and climate change (England) 2007
Global atmosphere	Column ozone levels at Lerwick and Camborne 1979–2006
Air quality	Annual levels of particles and ozone in the air (UK) 1987–2007 Days when air pollution is moderate or higher (UK) 1987–2007 Particulate (PM_{10}) emissions by source (UK) 1980–2006 Sulphur dioxide emissions by source, and targets (UK) 1980–2006 Nitrogen oxides emissions by source (UK) 1980–2006
Inland water	Biological river quality (England) 1990–2007 Chemical river quality (England) 1990–2007 Nitrate and phosphate concentrations in rivers (UK) 1995–2006 Abstractions for the public water supply from surface water and groundwater, by region (England and Wales) 2006 Water supply and leakage (England and Wales) 1994/5 and 2006/7 Drinking water quality (UK) 1995–2006 Properties at risk of flooding (England and Wales) 2006
Coastal and marine waters	Compliance with EC Bathing Water Directive mandatory and guideline standards (UK) 1988–2008 Compliance with EU Bathing Water Directive guideline standards (UK) 1995 and 2008 North Sea fish stocks and stocks of North East Atlantic mackerel (UK) 1964–2007 Fish stocks around the UK at full reproductive capacity and harvested sustainably (UK) 1990–2006
Radioactivity	Radioactive waste stock (UK) 1986–2007 Radioactive waste disposal (UK) 1997–2006 Discharges from the nuclear industry (UK) 1983–2005
Pollution incidents	Serious pollution incidents affecting water, air or land (England and Wales) 1993–2007 Serious pollution incident sources (England and Wales) 2000–2007
Waste and recycling	Household waste per person after recycling and composting (England) 1997/8–2007/8 Green and dry recycling rates for household waste (England) 1997/8–2007/8 Biodegradable municipal waste landfilled and targets (England) 2001/2–2007/8 Total waste landfilled and non-municipal waste to landfill (England) 2000/1–2006
Land	Agricultural and forestry land use (UK) 1996–2007 Agri-environment schemes (England) 1992–2007 Area of woodland (UK) 1924–2008 SSSI habitats in favourable or recovering condition by sector (England) 2003–2008 New homes built on previously developed land (England) 1989–2007
Wildlife	Status of priority species in the UK (UK) 2002–2005 Status of priority habitats in the UK (UK) 2002–2005 Populations of wild birds (UK) 1970–2007 Populations of wild birds by region (England) 1994–2006 Populations of butterflies (UK) 1976–2006 Spending on UK biodiversity (UK) 2000/1–2006/7 Spending on global biodiversity (UK) 2000/1–2006/7

Source: DEFRA 2008a

with biophysical data – it reports on public attitudes. It has more climate-change-related information. The US ROE does not give any information on radioactive wastes, but it provides more information on, non-greenhouse gas, emissions to the atmosphere.

19.1.3 Energy and materials flows

In Chapter 2, see Figure 2.1, we distinguished four roles for the environment in relation to the economy – source of resource inputs, waste sink, source of amenity services, life support system. As Tables 19.1 and 19.2 show, state of the environment reporting is primarily about the last three roles, and about renewable resources. Non-renewable resources stocks and flows generally do not get any coverage in state of the environment reporting – data on such stocks and flows are not usually regarded as environmental indicators. In this respect, Table 19.2 shows the UK's 2008 report to be somewhat unusual, providing data on energy consumption, and fuels used in electricity generation. This it did in connection with its coverage of the climate change theme, which was greater in 2008 than in previous years.

The main reason why official state of the environment reporting ignores energy and materials flows is that the organisation of government does not reflect the materials balance principle discussed in Chapter 2 – environmental extractions and insertions are typically dealt with by separate departments/agencies of government. Basically, the historical evolution of government has meant that 'environment' is about flora and fauna and pollution. Materials and energy extractions from the environment have been dealt with by government departments/agencies concerned with the industries that supply and use such as inputs to production. Such industries keep monetary accounts, and some aspects of their activities – expenditures and receipts – show up in standard national income accounts. It is perhaps for this reason that data, physical as well as monetary, relating to energy and materials tend to come under the rubric of 'accounting', rather than indicators. We will follow this convention and defer consideration of data on materials and energy extractions from the environment to the following sections on resource and environmental accounting.

19.2 Environmental accounting: theory

In a lecture delivered at the Washington DC 'think tank' Resources for the Future in 1992 (Solow, 1992, 1993), the Nobel laureate economist Robert Solow suggested that 'an innovation in social accounting practice could contribute to more rational debate and perhaps more rational action in the economics of non-renewable resources and the approach to a sustainable economy'. We use the title of his lecture as the heading to the next subsection. In it he outlined the basis in economic theory for his view that proper national income accounting would promote sustainability. As we have noted, many environmentalists share this view, as do many economists. For the economists, the theory outlined by Solow is the basis for their views on this matter. In this section we shall consider that theory, and the modifications to current national income accounting conventions that it is taken to imply. We shall also make the important point that there appears to be some misunderstanding of the theory and its implications for the ability of revised national income accounting conventions to promote sustainability. In the text here we shall try to tell the story in fairly intuitive terms. Appendices 19.1 and 19.2 tell the same story in mathematical terms. The theory to be considered here builds on the theory of natural resource use covered thus far in this Part of the book – Chapters 14, 15, 17 especially.

19.2.1 An almost practical step toward sustainability

We consider an economy that uses a non-renewable resource and human-made, reproducible, capital to produce output, which can be either consumed or added to the stock of capital. There is no technical progress. A sustainable path is one that involves constant utility for ever. Given that there is just one commodity produced and consumed, and that utility depends only on consumption, constant utility is the same as constant consumption. We are going to consider the question: what kind of economic behaviour is necessary for sustainability in this sense? There is, of course, a prior question, which is: can such an economy be sustainable? Given that the stock of the resource is finite, it is obvious that the answer to this question depends, as discussed in Chapters 3 and 14, on the possibilities for substitution in production as between the resource and reproducible capital. If those possibilities are such that sustainability is infeasible – as they would be if the production function was $Q = K^{\alpha}R^{\beta}$, $\alpha + \beta = 1$ and $\beta > \alpha$ – then following the rules for economic behaviour that are the answer to the first question could not deliver sustainability. Those rules are necessary but not sufficient conditions for sustainability. We return to the question of feasibility at the end of this section.

As set out by Solow, the theory involves two 'key propositions'. The first is that 'properly defined net national product'

> measures the maximum current level of consumer satisfaction that can be sustained forever

and is therefore

> a measure of sustainable income given the state of the economy

The second proposition is that

> Properly defined and properly calculated, this year's net national product can always be regarded as this year's interest on society's total stock of capital

where the total stock of capital includes both reproducible capital and the resource stock. When these two propositions are put together, we get the rule for economic behaviour that gives sustainability. It is to maintain society's total stock of capital intact, by consuming only the interest on that capital. This implies adding to the stock of reproducible capital an amount equal to the depreciation of the resource stock – which is Hartwick's rule, discussed in Chapters 3 and 14 – where the depreciation of the resource stock is measured by the Hotelling rent arising in its extraction.

In his lecture, Solow was careful to state, several times, a caveat that attends these propositions as guides to policy in an actual economy. This is that the 'right prices' are used to value the capital stock and the resource stock. Note that without prices we could not add together the stocks of reproducible capital and the resource to get a figure for 'society's total wealth'. In order to be 'right', the prices must be, as Solow puts it, such that they 'make full allowance even for the distant future, and will even take account of how each future generation will take account of its future'. The theory ensures that the prices do this by working with a model in which there is a single representative agent with perfect future knowledge, who works out and follows a plan for consumption, investment and resource depletion on the basis of maximising the discounted sum of future utilities subject to the constraints imposed by the availability of the resource and the need to forgo consumption in order to invest in reproducible capital. Such a model is set out in Appendix 19.1, and was previously considered in Chapters 3, 14 and 15.

The justification for using such a model to think about these questions is that, given some very strong assumptions about agents' foresight and institutions, competitive markets would produce the same price behaviour. The model shows, for example, that the resource price is required to evolve according to Hotelling's rule, and, given strong assumptions, it can be shown that resource prices determined in competitive markets will follow the same rule. Solow is absolutely explicit about the relationship between actual market prices and the 'right prices' for guiding the economic behaviour that is necessary for sustainability:

> This story makes it obvious that everyday market prices can make no claim to embody that kind of foreknowledge. Least of all could the prices of natural resource products, which are famous for their volatility, have this property; but one could entertain legitimate doubts about other prices, too. The hope has to be that a careful attempt to average out speculative movements and to correct for the other imperfections I listed earlier would yield adjusted prices that might serve as a rough approximation to the theoretically correct ones. We act as if that were true in other contexts. The important hedge is not to claim too much.

Unfortunately, in their enthusiasm to use economic theory to promote sustainability, some economists do not explicitly qualify their contributions to policy analysis with 'the important hedge'.

There is, as set out in Appendix 19.1 and discussed below, a further 'hedge' of some importance, not made explicit in Solow's lecture, and which tends to be glossed over in much of the literature. This is the fact that, even within the context of the representative agent model itself, the prices may not be 'right'. The 'right' prices are those which go with a constant consumption path. However, the representative agent will not necessarily choose a constant consumption path, unless constrained to do so. Hence, the prices ruling along the optimal path in such a model will not be the correct prices to use for the implementation of Hartwick's rule in pursuit of sustainability, unless it so happens that the representative agent's optimal path is one with constant consumption.

What has all this got to do with environmental accounting? What is Solow's 'almost practical step'? It is the idea that if at a point in time we knew what sustainable income for the economy was we would know whether or not we were behaving in the interests of the future. Consumption in excess of sustainable income would indicate that we were not, while consumption equal to or less than sustainable income would indicate that we were. The step is 'almost practical' because of the need to use not currently observable market prices, but the 'right' prices. Given the obvious difficulty of figuring the 'right' prices, it might be argued that 'An impracticable step toward sustainability?' would have been a better title for Solow's 1992 lecture.

19.2.2 A resource owner in a competitive economy

The idea that to behave sustainably involves keeping wealth intact by consuming just the interest income on that wealth has considerable intuitive appeal. We can make that appeal explicit by considering the situation of a resource owner in a competitive economy. Doing this will also serve to provide some insight into how the Hartwick rule works when it does, and some basis for a further discussion of the caveats noted above.

Consider, then, an individual who owns an oil deposit and sells extraction permits to a company in the oil production business. The individual pays the proceeds from permit sales into his or her bank account, from which is paid his or her expenditure on consumption. Let us use here the following notation:

B is the size of the bank account, units £s
C is consumption expenditure, units £s
W is total wealth, units £s
R is the total of permit sales, units tonnes
X is the size of the remaining stock of mineral, units tonnes
h is the price of a permit, £s per tonne
V is the value of the mine, units £s
i is the interest rate, assumed constant over time

Let us also use $t - 1$ to denote the first day of the relevant period of time, say a year, and t to denote the last day of the period. At $t - 1$ the mine owner sells permits and banks the revenue. At t he or she writes a cheque on the bank account to pay for his or her consumption during the period. While this construction is somewhat special it serves to make what is going on clear. In this context, considering, as we shall, an infinite time horizon and the question of constant consumption by an individual for ever is obviously rather strange. Individuals do not live for ever. However, pretending that they do, or at least that they behave as if they do by treating their heirs as simple extensions of themselves, is not uncommon in economics, and does serve to generate some useful insights.

The behaviour over time of B is given by

$$B_t = (1 + i)B_{t-1} + (1 + i)h_{t-1}R_{t-1} - C_t \tag{19.1}$$

because B_{t-1} is the principal at the start of the year, to which is added, to earn interest over the year, the proceeds from permit sales at the start of the year. Equation 19.1 can be written as

$$B_t - B_{t-1} = iB_{t-1} + (1 + i)h_{t-1}R_{t-1} - C_t \tag{19.2}$$

At t the value of the mine is given by the permit price at t multiplied by the amount of oil remaining, which is the amount remaining at the start of the period less the amount for which permits were sold at the start of the period. That is:

$$V_t = h_t(X_{t-1} - R_{t-1}) \tag{19.3}$$

The price of an extraction permit in a competitive economy will be the difference between the marginal cost of extraction and the price for which extracted oil sells; that is, the Hotelling rent. That is why we have used 'h' here as the symbol for the price of an extraction permit. Again given a competitive economy, we know from Chapter 15 that Hotelling's rule governs the behaviour of rent, and hence the price of extraction permits, over time so that

$$h_t = (1 + i)h_{t-1}$$

and substituting in equation 19.3 gives

$$\begin{aligned} V_t &= (1 + i)h_{t-1}(X_{t-1} - R_{t-1}) \\ &= (1 + i)(h_{t-1}X_{t-1} - h_{t-1}R_{t-1}) \end{aligned}$$

or

$$V_t = (1 + i)(V_{t-1} - h_{t-1}R_{t-1}) \tag{19.4}$$

from which we get

$$V_t - V_{t-1} = iV_{t-1} - (1 + i)h_{t-1}R_{t-1} \tag{19.5}$$

The individual's wealth is just the sum of the bank deposit and the value of the mine:

$$W_t = B_t + V_t$$

so that the change in wealth over a period is:

$$W_t - W_{t-1} = (B_t - B_{t-1}) + (V_t - V_{t-1}) \tag{19.6}$$

Substituting in equation 19.6 from equations 19.2 and 19.5 gives

$$\begin{aligned} W_t - W_{t-1} = iB_{t-1} &+ (1 + i)h_{t-1}R_{t-1} - C_t + iV_{t-1} \\ &- (1 + i)h_{t-1}R_{t-1} \end{aligned}$$

or

$$W_t - W_{t-1} = iB_{t-1} + iV_{t-1} - C_t \tag{19.7}$$

which is

$$W_t - W_{t-1} = iW_{t-1} - C_t \tag{19.8}$$

Now, for constant wealth, $W_t - W_{t-1} = 0$, we get from equation 19.8

$$C_t = iW_{t-1} \tag{19.9}$$

so that if a period's consumption is equal to the interest earned on total wealth at the start of the period, wealth will be the same at the end of the period as at the start. Further, equation 19.9 holds for all t and $t - 1$, for all periods, so that if we use the subscript 0 for the start of some initial period

$$C_t = iW_0 \tag{19.10}$$

will clearly be the maximum constant consumption level for all subsequent periods. Readers who are unconvinced that iW_0 is the largest possible constant consumption stream can convince themselves that this is the case by some numerical experiments. A numerical example on which such experiments could be based is given in chapter 9 of Common (1996).

Given the result that the present value of x for ever is x/i (see Chapter 11) the present value of the consumption stream iW_0 for ever is

$$W_0^* = W_0 \tag{19.11}$$

so that wealth as the current value of total assets and wealth as the present value of the largest future constant consumption level that is indefinitely sustainable are the same.

For this individual a period's income, Y, is given by the interest payment on the bank deposit plus the revenue from permit sales and the interest earned thereon:

$$Y_t = iB_{t-1} + (1 + i)h_{t-1}R_{t-1} \tag{19.12}$$

Equation 19.7 for $W_t - W_{t-1} = 0$ gives

$$C_t = iB_{t-1} + iV_{t-1}$$

and if we define investment, I, as the difference between income and consumption we have

$$\begin{aligned} I_t = Y_t - C_t &= iB_{t-1} + (1 + i)h_{t-1}R_{t-1} - iB_{t-1} - iV_{t-1} \\ &= (1 + i)h_{t-1}R_{t-1} - iV_{t-1} \end{aligned} \tag{19.13}$$

From equation 19.5, this can be written as

$$I_t = -(V_t - V_{t-1}) \tag{19.14}$$

which says that the individual is investing an amount equal to the depreciation of the mine. This is Hartwick's rule applied to this individual – investing in his or her reproducible capital, the bank account, in every period an amount equal to the depreciation of his or her resource stock, the oil deposit. The depreciation of the oil deposit is simply the reduction in its value over the period on account of the reduced size of the resource stock. Note that equation 19.14 can also be read as saying that net investment – that is, investment less depreciation – is zero when wealth is maintained intact.

A widely used definition of 'sustainable income' is that it is the amount that can be consumed during a period without reducing wealth. Here it follows immediately from the preceding discussion that with $Y_{sus,t}$ for the individual's sustainable income for the period starting on $t - 1$ and ending on t, it is:

$$Y_{sus,t} = iW_{t-1} \tag{19.15}$$

Recall that Solow stated that properly measured net national product, or income, is both the interest on wealth and the level of consumption that can be maintained for ever. Equation 19.15 gives sustainable income for the individual as the interest on wealth. We have already established in this context (equation 19.9) that this is a level of consumption that can be maintained for ever.

All of this is the basic result set out by Solow as discussed in the previous subsection – for sustainable consumption, maintain wealth intact by consuming just the interest on the constant wealth – but here it is shown to work for a non-renewable-resource-stock-owning individual in a competitive economy with a constant interest rate rather than for an economy as a whole.[2] We look at the transferability of the result to an economy in the next subsection, but before doing that there are some further points to be made about the situation of an individual.

The first is to note the key role of the efficiency condition that the proportional rate of increase in rent is equal to the single ruling interest rate. If we have $h_t = (1 + b)h_{t-1}$ rather than $h_t = (1 + i)h_{t-1}$, then we cannot derive equation 19.7.

We have shown that by consuming just the interest on wealth an individual resource-stock owner achieves the highest sustainable level of consumption. We have not shown that such an individual would choose such a consumption pattern. In fact an individual would do so only in special circumstances, as we now show. There is a substantial and technically sophisticated literature on the choice of intertemporal consumption plans by individuals, but for our purposes a very simple formulation of the problem will suffice. We assume that the problem of choosing a consumption plan can be represented as

$$\text{Max} \int_0^\infty U(C_t)\mathrm{e}^{-\rho t}\mathrm{d}t$$

$$\text{subject to } \mathrm{d}W/\mathrm{d}t = iW_t - C_t$$

where the notation is as before, but we have introduced the symbol ρ for the rate (assumed constant) at which the individual discounts future utility. For this problem, we get from the current-value Hamiltonian necessary conditions which include

$$\partial H_t/\partial C_t = U_{C_t} - \lambda_t = 0$$

and

$$\mathrm{d}\lambda/\mathrm{d}t - \rho\lambda_t = -\partial H_t/\partial W_t = -\lambda_t i$$

where from the second condition we can write

$$\mathrm{d}\lambda/\mathrm{d}t = (\rho - i)\lambda_t$$

and we have, on standard assumptions about diminishing marginal utility,

$$\rho = i \rightarrow \mathrm{d}\lambda/\mathrm{d}t = 0 \rightarrow U_{Ct} \text{ constant, } C_t \text{ constant}$$

$$\rho > i \rightarrow \mathrm{d}\lambda/\mathrm{d}t > 0 \rightarrow U_{Ct} \text{ increasing, } C_t \text{ decreasing}$$

$$\rho < i \rightarrow \mathrm{d}\lambda/\mathrm{d}t < 0 \rightarrow U_{Ct} \text{ decreasing, } C_t \text{ increasing}$$

Thus, we see that the individual will choose constant consumption as his or her optimal plan only if their intertemporal utility discount rate is equal to the interest rate.

Suppose that our individual started out with C_t increasing and then decided for some reason at the start of period $T - 1$ to T decided to switch to C_t constant. Given the foregoing it should be clear that the individual would thereafter be acting to consume the interest on the wealth at $T - 1$ and maintaining that wealth intact and sustaining constant consumption for ever, but that the wealth maintained intact would be less than the individual's initial wealth and the constant indefinitely sustainable consumption level would be lower than if such behaviour had been adopted at the outset. Again, the reader who wishes to confirm these points can do so by simple numerical experimentation.

There is another point here that can also be confirmed in that way. We have not yet mentioned the eventual exhaustion of the oil deposit, and what happens when that occurs. In fact, in regard to consumption and wealth, nothing happens. By the time the oil is exhausted, given the behaviour from the outset that keeps wealth constant, the entire initial value of the oil stock will have been transferred to the bank deposit, and our individual can continue to have constant consumption for ever as the interest on the bank deposit at the same level as initially.

[2] Note that if the interest rate r is not constant, sustainable consumption would not maintain wealth intact. Wealth would have to move in the opposite direction to any change in the interest rate, so that the product rW remains constant.

Finally, we should note that, in order to establish a reference point for talking about economies as opposed to individuals, we have thus far ignored one area of opportunity open to an individual. We have assumed that the individual's consumption opportunities over time are given solely by the way he or she manages the asset portfolio, which comprises the bank account and the mine. In fact, an individual can borrow to alter his or her consumption path from that given by the asset portfolio, incurring debts for later repayment from the income stream that it generates. We considered behaviour in loanable funds markets in Chapter 11. It was shown there that, given standard assumptions, the individual would manage assets so as to maximise wealth, and then choose a consumption path reflecting his or her intertemporal preferences and the opportunities available by borrowing and repaying. There is, that is, a 'separation theorem' which shows that the problems of asset portfolio management and consumption planning can be treated sequentially – first maximise wealth ignoring intertemporal preferences, then arrange consumption over time subject to the constraint arising from maximised wealth. We make this point to emphasise that our discussion here has been intended only as a means of providing some sense of economists' basic way of thinking about sustainability, rather than as a full account of intertemporal consumption planning by an optimising individual.[3]

19.2.3 Consumption, income and wealth for an economy

We now consider an economy which uses reproducible capital and a non-renewable resource to produce output, which can be either consumed or saved and added to the capital stock. The first point to be made is that there is an important distinction between an open economy, which trades with other economies, and a closed economy where there is no foreign trade. If we assume that an open economy is 'small', so that it takes world prices for traded goods as given, that there is complete freedom of international capital movements (with respect to which the economy is also 'small') and that all markets are competitive, then the situation for an open economy is essentially as set out above for an individual. We will return to this point in discussing two essential differences between a resource-owning individual and a resource-exploiting economy in relation to sustainability as constant consumption for ever. Note that the global economy is a closed economy, and that the sustainability problem is really a global problem.

The first essential difference primarily concerns the feasibility of constant consumption for ever. Equation 19.12 is the production function for the individual's income. It is linear, implying that the bank account and the oil deposit are perfect substitutes in the production of income, and that the oil deposit is not essential in the production of income. As we noted above, for the individual mine owner exhaustion of the mine does not, if behaviour has previously been such as to maintain wealth intact, imply any reduction in wealth or sustainable consumption. In the economics literature on sustainability, it is not generally regarded as appropriate to assume for an economy that the resource is inessential in production. Note, however, that most of the literature is concerned with a closed economy. For a small open economy which can export the resource and invest in overseas assets the situation is essentially as for an individual and an income production function like equation 19.12 would be appropriate. For a closed economy, it can be shown that even where the resource is essential, sustainability as constant consumption may be feasible. This is, as already noted, the case where the production function is

$$Q_t = K_t^{\alpha} R_t^{\beta} : \alpha + \beta = 1 \text{ and } \beta < \alpha \tag{19.16}$$

where K_t is the stock of reproducible capital and R_t is the resource use at time t.

The second essential difference concerns the behavioural rule that will give sustainability, if it is feasible. The point here is that whereas for an individual in a competitive economy, or for a small open economy in a competitive world economy, prices are exogenous and unaffected by the behaviour of

[3] Most standard intermediate and advanced microeconomics texts provide an analysis of the individual intertemporal utility maximisation: see, for examples, Hirshleifer (1980) and Deaton and Muellbauer (1980). Chapter 16 in Hirshleifer (1980) discusses the separation theorem, on which see also chapter 6 of Common (1996).

the individual or small open economy, for a closed economy prices are endogenous and depend upon the economy's behaviour. Also, for the individual analysed in the previous subsection, the marginal product of the bank account in producing income is constant, and equal to i, the single interest rate ruling throughout the economy. For a closed economy with equation 19.16 as its production function, on the other hand, the behaviour of the marginal product of capital over time depends upon the time paths chosen for K_t and R_t, as does the marginal product of resource use. Analysing properly the full implications of the endogeneity of prices and rates of return on assets is difficult, and we will not attempt it here. Some analysis is provided in Appendix 19.1. We can, however, use Figure 19.1 to show some of the results.

In Figure 19.1 C_t^o is the optimal path for consumption from a representative agent model of a closed economy, where the integral of the discounted utility of consumption is maximised, subject to the constraints involving resource use and the allocation of output as between consumption and capital accumulation.[4] This is essentially the model considered in Chapters 3 and 14; see also Appendix 19.1. C_o^S is the constant consumption level that could be maintained for ever if at the outset the agent went for the highest feasible level of constant consumption, rather than the optimal path C_t^O; and C_t^S gives the time path under the optimal plan for the maximum level of consumption that would be indefinitely sustainable at each date. At T, for example, C_T^O is optimal consumption and C_T^S is the maximum constant level of consumption sustainable after T, given that the optimal plan is followed until T.

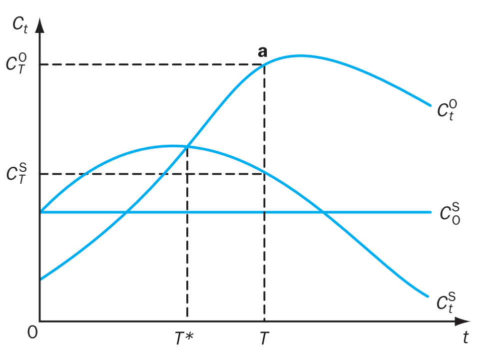


Figure 19.1 Optimal and sustainable consumption paths

Now, the theory appealed to by Solow and discussed above has been interpreted as having the implication that if an economy had been following the optimal path and were at the point a on C_t^O the ruling prices and interest rate could be used with the stock of capital and of the resource to compute wealth and sustainable income for which the corresponding future constant consumption level would be C_T^O. In fact, as Figure 19.2 shows, C_T^O would not be sustainable at T, given that C_t^O had been followed to that time. The maximum constant consumption level that could be indefinitely sustained forward from T, given that C_t^O had been followed until T, is C_T^S. Note that it is not being asserted that using prices and quantities from the optimal path will always overstate wealth, sustainable income and future constant consumption. To the left of T^* C_T^S is greater than C_t^O. The point is that, in general, using the prices and quantities that go with the optimal path will give incorrect signals regarding the level of sustainable income and constant future consumption as interest on wealth. To get the right signals at time T it would be necessary to use the prices and quantities that would hold at T on the path C_t^S. Note that both the efficiency condition (rent increasing proportionately at the rate of interest) and the Hartwick rule (zero total net investment) hold along C_o^S, given that the prices that go with that time path for consumption are used in stating them.

As we have already noted, in the case of an individual it is also true that the optimal consumption plan will, generally, not involve maximum constant consumption. And, to the extent that it does not, an individual who follows it initially will subsequently have stocks of assets that are different from those that would exist if the maximum constant consumption path had been followed. However, the prices and interest rate facing the individual, being independent of his or her behaviour, will not be affected. He or she could, should at some point he or she wish to follow a constant-consumption path, use the ruling prices and interest rate to work out the maximum

[4] Figure 19.1 is based on figure 4 in Pezzey (1997).

constant level of consumption possible, given the actual asset portfolio.

In discussing the case of the closed economy here we have been looking at a model economy tracking the optimal path, which is usually taken to being equivalent to thinking about the path that a fully competitive perfect-foresight economy would track. Of course, as emphasised by Solow, actual economies do not involve perfect foresight and competition. The model economy considered is special in a number of other ways also – it exploits just one non-renewable resource, there is no population growth and no technical progress. Models without such restrictions have been examined in the literature, and references are provided in the Further Reading section at the end of the chapter. We have focused here on this very simple model in order to highlight in it a point that applies generally – measuring sustainable income, and hence future constant consumption possibilities, requires using the prices that go with sustainability to measure total wealth. These are not, generally, the prices that obtain along the path that a competitive perfect-foresight economy would track, and are not the prices that we observe in actual economies. These 'important hedges' are often overlooked, with the result that the prospects for actually measuring sustainable income for an actual economy are frequently oversold.

In saying this, it is implied that there is a sustainable income to be measured, that constant consumption for ever is feasible. As we have noted, this is not assured. There is an extensive literature on feasibility conditions in the simple model considered here, and on extensions to encompass multiple resource inputs (renewable and non-renewable), population growth and technical progress. Again, we refer the reader to the Further Reading section, and move to considering the adjustments to standard measures of national income that this sort of economic theory, which we shall refer to as 'capital theory' in what follows, suggests are required.

19.2.4 Measuring national income

The end purpose of economic activity is the satisfaction of human wants and needs, i.e. consumption. However, the most widely used measure of how well an economy performed during a year is its national income, which comprises both consumption and investment. What is the theoretical justification for including investment in a measure of national economic performance? In essence, the answer to the question is that current investment contributes to future consumption.

19.2.4.1 The environment ignored

Leaving aside matters environmental for the moment, the model used most widely by economic theorists to investigate the proper measure of an economy's performance is one where a single commodity is produced, using just labour and capital, which can be consumed or added to the capital stock. To simplify, it is also assumed that capital does not depreciate in use, that there is no government economic activity, no foreign trade, and that the population is constant. Given the last of these, nothing essential is lost by assuming that production just uses capital. In terms of production possibilities, this is the model that we first encountered in Chapter 3 (see also Chapter 11) and when looking at optimal growth by considering the problem of the maximisation of

$$\int_0^\infty U(C_t)e^{-\rho t}dt$$

subject to

$$\dot{K} = Q(K_t) - C_t$$

with U for utility, C for consumption, Q for output, K for capital and with ρ for the utility discount rate. This problem can be understood either to be that facing an immortal representative individual or that facing a planner operating on behalf of society.

When looking at this problem before we have been interested in the solution in terms of what the optimising time profiles for the variables look like, and what conditions hold along the optimal solution. Now we are interested in whether there is some function of current levels of the variables that gives a single valued measure of performance in terms of the objective function. As shown in Appendix 19.1, there is and it is

$$U(C_t) + U_C\dot{K}_t$$

where U_C is the marginal utility of consumption. If we assume that the utility function is linear so that $U(C_t) = U_C C_t$, then, using $I_t = \dot{K}_t$ we can write this as

$$U_C C_t + U_C I_t$$

which is a performance measure in units which are utils. Dividing through by U_C converts this to $C_t + I_t$, which is an expression in units of the single commodity, and

$$\mathrm{NDP}_t = C_t + I_t \tag{19.17}$$

is a performance measure purged of utils.

Here, NDP stands for Net Domestic Product. The reason for this terminology is that prior to the working out of this bit of theory it had become established practice in national income accounting to treat $C_t + I_t$ (with I_t as net investment) as the proper measure of aggregate economic performance, and because it was there called Net Domestic Product.

In national income accounting, 'income' and 'product' are just two different ways of looking at the same thing. NDP properly measures income as well as output. We could just as well use NNI, for net national income, to refer to consumption plus investment. In much of the literature, NDP is the symbol while national income, rather than national product, is the form of words. Thus, it is typical to refer to 'sustainable income', rather than to 'sustainable product', but to use NDP as the symbol for sustainable income.

Define sustainable income as the maximum that can be consumed without reducing the capital stock by consuming some of it, and recall that I_t is net investment. From equation 19.17

$$\mathrm{NDP}_t - C_t = I_t$$

so that

$$C_t \geq \mathrm{NDP}_t$$

means

$$I_t \leq 0$$

If C_t exceeds NDP_t then net investment is negative, and so $K_{t+1} \leq K_t$, which by virtue of $Q_t = Q(K_t)$ means that Q_{t+1} will be less than Q_t.

19.2.4.2 Taking account of the environment

Theoretical arguments about how sustainable income should be measured when the interdependence of the economic and the environment is recognised have been developed in terms of modifications to equation 19.17. These are derived from models that all have the same structure and nature, which is that of the optimisation model considered in the previous subsection, except in regard to what is assumed about the way the economy relates to the environment. A number of such models are presented and analysed in Appendix 19.2. Here we shall briefly review some of the results reported there. Basically what we find is that taking account of the environment when measuring sustainable national income requires adjustments to NDP, as defined by equation 19.17, which register changes in the state of the environment as it affects current and future consumption.

For the model that is the basis for Figure 19.1, where production requires inputs of a single nonrenewable resource as well as reproducible capital, and using EDP (environmentally adjusted domestic product) for sustainable income, the result is that

$$\mathrm{EDP}_t = \mathrm{NDP}_t - Q_{Rt}R_t = \mathrm{NDP}_t - h_t R_t \tag{19.18}$$

where Q_{Rt} is the marginal product of the resource in production, R_t is the amount used, and h_t is, as previously in this section, the Hotelling rent. In this model, resource extraction is costless, so that Hotelling rent is equal to marginal product. The second term on the right-hand side of equation 19.18 is the depreciation of the resource stock. Two points need to be made here.

First, if we substitute equation 19.17 into equation 19.18 we get

$$\mathrm{EDP}_t = C_t + I_t - h_t R_t$$

so that if total net investment is zero – investment in reproducible capital equals resource depreciation, the Hartwick rule – we have consumption equal to sustainable income and, given the caveats of the previous subsection, constant wealth.

The second point concerns the interpretation of Hotelling rent times resource use as depreciation of the resource stock. Earlier, at equation 19.5, we gave depreciation in the value of the mine as:

$$V_t - V_{t-1} = iV_{t-1} - (1 + i)h_{t-1}R_{t-1}$$

Rearranging 19.5 we get depreciation as the change in the value of the mine as

$$V_t/(1+i) - V_{t-1} = h_{t-1}R_{t-1} \tag{19.19}$$

where the right-hand side refers to the start of a period, so that the value of the mine at the end of the period, V_t, has to be discounted by $(1 + i)$ on the left-hand side for comparability. In both discrete and continuous time, depreciation of the resource stock/mine is equal to Hotelling rent times the amount extracted.

In the model which is the basis for Figure 19.1, and for which equation 19.18 is derived, resource extraction is costless, and there is no exploration activity that can increase the size of the known resource stock. In a model economy where resource extraction involves cost, and new known reserves can be established at some cost, we find that

$$\begin{aligned} \text{EDP}_t &= \text{NDP}_t - (Q_{Rt} - G_{Rt})(R_t - N_t) \\ &= \text{NDP}_t - h_t(R_t - N_t) \end{aligned} \tag{19.20}$$

where Q_{Rt} is the marginal product of the resource, G_{Rt} is marginal extraction cost, and N_t is additions to the known stock as the result of exploration. Note that where extraction is costly, Hotelling rent is the difference between the marginal product of the resource and its marginal cost of extraction.

Suppose that a renewable resource rather than a non-renewable resource is used in production. Then we find that

$$\begin{aligned} \text{EDP}_t &= \text{NDP}_t - (Q_{Rt} - G_{Rt})(R_t - F\{S_t\}) \\ &= \text{NDP}_t - h_t(R_t - F\{S_t\}) \end{aligned} \tag{19.21}$$

where G_{Rt} is the marginal cost of harvesting, and $F\{S_t\}$ is the growth function for the resource stock, where S_t is the stock size. Note that equation 19.21 has exactly the same structure as equation 19.20, with $F\{S_t\}$ playing the role in 19.21 that N_t plays in 19.20. Note also that for sustainable yield exploitation of the renewable resource, $R_t = F\{S_t\}$, there is no depreciation to account for, and $\text{EDP}_t = \text{NDP}_t$.

While some renewable resources are solely of economic interest as an input to production, some are also of direct interest to consumers. An example would be some tree species which is harvested and used in the production of commodities such as paper, and which as standing timber is a source of recreational services. In such a case we find that

$$\text{EDP}_t = \text{NDP}_t + (U_{St}/U_{Ct})S_t - h_t(R_t - F\{S_t\}) \tag{19.22}$$

where U_{St} is the marginal utility of standing trees and U_{Ct} is the marginal utility of produced commodity consumption. As compared with equation 19.21 we now have an additional adjustment to make to net national income as conventionally measured. We have written this adjustment for the amenity value of standing timber valuing it using a ratio of marginal utilities because, typically, there will be no market price that we can observe for the amenity services of standing timber. If we want to measure sustainable income taking account of such amenity services, then we cannot rely on market prices. We could, in principle, think in terms of getting some kind of price to use with S_t from the methods discussed in Chapter 12, where we also discussed the problems that attend such methods.

More fundamentally, there is the question of whether there should be such an adjustment to NDP_t for any particular tree species. In the model which leads to equation 19.22 it is assumed that standing timber yields services to consumers. In the model which leads to equation 19.21 this assumption is not made. The different prescriptions about adjustments to NDP_t arise from different models about how the economy relates to the environment. For any particular tree species we could, in principle, decide which is the appropriate model by using the methods of Chapter 12 to test for $U_S = 0$. While this is true, given many renewable resources which *a priori* could have direct utility it does imply a large research agenda for actually doing environmental accounting. In the absence of such empirical resolution, the prescriptions from capital theory for adjusting conventional income measurement to account for the environment are dependent on the assumptions embodied in the model.

As shown in Appendix 19.2, the point here also applies when we start to consider adjustments on account of the environmental deterioration due to emissions arising in production. What capital theory tells us about how to do environmental accounting and measure sustainable income depends on the model of economy–environment interdependence that is used. Given the current state of knowledge, there is no unique and generally agreed model.

Given the uncertainty, as ignorance, that is central to the sustainability problem, it is unlikely that there will ever be such a model. It would, in any case, be a very complicated model, and unlikely to generate simple prescriptions for national income accounting purposes. While capital theory can provide some general insights, it cannot provide generally agreed definitive rules for practicing national income accountants to follow. Further, the pricing caveats discussed in the context of the simplest model with production using a costlessly extracted non-renewable resource carry through to all the more complex models. And, as we shall see in the next section, even where the theory offers clear and unambiguous prescriptions over a limited area of the total problem, as with non-renewable resource depletion, implementation remains problematic.

19.3 Environmental accounting: practice

In this section we consider environmental accounting from a perspective which is that of a national income statistician rather than an economic theorist. We begin with some observations on current national income accounting conventions, and their deficiencies in relation to matters environmental as argued by many commentators. We then look at proposals emanating from the United Nations for addressing such concerns. The second subsection looks at the different ways in which non-renewable resource depreciation can be, and has been, measured in practice. Then we look at two (partial) implementations of the UN proposals. We finish by looking at some attempts to produce a measure which better reflects a nation's economic progress, or the lack of it, when due account is taken of its environmental impacts.

Current national income accounting conventions actually produce a variety of measures relating to national income. The most widely used are Gross National Product (GNP) and Gross Domestic Product (GDP). The difference between GNP and GDP is not great for most economies, and its origins are not very relevant to our central concerns here. We shall conduct our discussion by referring to GDP.[5] The conventions now used for GDP measurement have their origin in the information requirements for management of the macroeconomy. For this purpose, what is needed is a measure of the total demand for the outputs of produced commodities. Given that GDP measures total demand, it also measures the output produced to meet that demand, and GDP has come to be seen as a measure of economic performance, or welfare. Indeed, for many commentators it has effectively become *the* performance/welfare indicator, notwithstanding that economists have long been aware of many ways in which it is, in practice, a very poor performance/welfare indicator, even leaving aside environmental considerations.

GDP can be measured in three ways. First, GDP is the total output sold by firms measured by value added. In measuring national income, purchases of intermediate goods are netted out, as discussed in Chapter 8. Second, GDP is the sum of the incomes earned by persons in the economy. This is the most obvious rationale for calling GDP 'national income'. The sum of incomes is equal to the value of total output produced by firms by virtue of the convention that output is measured in terms of value added. Third, GDP is total expenditure by individuals on consumption plus expenditure by firms on items of capital equipment, investment. Given these conventions, each way of measuring GDP will, in principle, produce the same numerical result. The value-added measure of firms' total output equals the incomes generated in firms equals total expenditure on non-intermediate goods.

It is universally agreed that, leaving aside environmental considerations, the proper measure of national income for purposes of monitoring national economic performance and welfare is Net Domestic Product (NDP). This is GDP less that part of it required to make good the depreciation of reproducible

[5] The national income accounting conventions are discussed in most macroeconomics texts. Beckerman (1980) provides a fuller discussion of the conventions than most such texts, and looks at the principles underlying them. Usher (1980) gives a very thorough discussion of the use of national income accounting data to measure economic growth, and contains an early discussion of adjustments for resource depletion and environmental deterioration. National statistical agencies publish detailed guides to the practices followed in their own accounts and publications; see, for example, Office for National Statistics (1998) for the UK.

capital as it is used in production. In principle, depreciation for a period is measured as the reduction in the value of the economy's existing stock of capital equipment over that period, on account of its use in production. In fact, GDP is much more widely used than NDP. The reason for this is that it is difficult to measure the depreciation of capital equipment accurately. National income statisticians prefer a number which is an accurate measure of an admittedly unsatisfactory concept to an inaccurate measure of a more satisfactory concept. This needs to be kept in mind when considering proposals for modifying national income measurement so as to account for the depreciation of environmental assets.

As noted in the introduction to this chapter, environmentally driven criticism of current accounting conventions focuses on three areas: depletion of natural resources, environmental degradation and defensive expenditure.

As regards natural-resource depletion, the widely agreed principle is that stocks of natural resources such as oil and gas reserves, stocks of fish, and so on should be treated in the same way as stocks of human-made capital, so that a deduction should be made to allow for the depletion or consumption of these natural resources as they are used in production – that is, their depreciation. In this regard, there is a distinction between resources that yield monetised flows (such as commercial forests, exploited oils and minerals, and so on) and those that yield non-monetised benefits (such as fresh air, lakes and oceans, and similar natural resources to which there are no exclusive property rights). In principle, the depreciation of the former ought to be observable in market data, while this will not be true of the latter. Where renewable resources are not traded in markets, or where they are exploited on an open-access basis, it is clearly going to be difficult to get firm data relating to depreciation. As well as the problem of valuation, there are often problems of physical measurement, given that there are no incentives for private measurement activity, as there are in the case of traded resources where exclusive property rights exist.

Degradation occurs when there is a decline in the quality of the natural environment, in particular of air, water and land quality. As with renewable and non-renewable resources, land, air and water can be viewed as assets, the degradation of which should be treated as depreciation and accounted for in the same way as depletion of reproducible capital. At the level of theory, there is no difference between this case and that of the depletion of natural resources. However, as a practical matter it is not always obvious how degradation should be defined and, even if satisfactorily defined, how it should be valued. An approach that has been suggested is to establish certain desirable quality standards, and then to measure degradation as the deviation from these quality levels. The value of the degradation can then be calculated as the cost of making good the degradation that has occurred or the cost of achieving the targeted quality standards. However, there is clearly the possibility of an arbitrary element in this since quality standards may be set which are higher than would occur in the 'natural' environment. It is unlikely that the quality standards established would be those which correspond to the efficient level of abatement – that is, where the marginal social cost of the pollution equals the marginal abatement cost. If the use of the costs of achieving standards is considered inappropriate, alternative methods of valuing degradation must be sought. Willingness to pay (WTP) to avoid the degradation, or to make it good, has been proposed. The assessment of WTP in relation to the natural environment was considered in Chapter 12. Leaving aside the problems discussed there, from a national accounting standpoint there is the difficulty that WTP includes consumers' surplus whereas the standard components of the national accounts are valued using market prices.

Expenditures that are expressly designed to prevent degradation or to counteract the effects of degradation that has already taken place – so-called defensive expenditures – will be included in GDP as currently measured. Expenditure incurred by producers – for example, on waste treatment by enterprises – will be reflected in product prices but will not be separately identifiable in the national accounts. Expenditure by households, government or non-profit-making institutions, or capital expenditure by enterprises, will be included on the expenditure side of GDP and should in principle be separately identifiable in the accounts currently produced. As noted, some commentators argue that such defensive expenditures should be deducted from GDP as now measured to arrive at a proper measure of national income. As a practical matter, quite apart from the difficulty of measuring defensive expenditure, it can be argued that there is no reason why defensive environmental expenditure should be treated differently from other forms of defensive expenditure, such as expenditure on armed forces, preventive medicine, policing and so on. A consistent approach to defensive expenditure would require major changes in the measurement of national income, beyond those required on environmental grounds. Some commentators argue that given the difficulties that such an approach would face, the construction of a measure of sustainable economic welfare should, rather than involve adjustments to national income, start somewhere else. We discuss some efforts of this nature, in the context of the construction of an 'index of sustainable economic welfare' in the final section of the chapter.

Before looking at modifications to national income accounting conventions, we should note that, leaving aside matters environmental, the measurement of national income is not an exact science. National income statisticians typically refer to the numbers that they produce for GDP, for example, as 'estimates'. Some idea of the inherent imprecision in national income accounting can be gained by looking at how the GDP estimate for a given year changes over time with revisions on account of new information becoming available. The UK's Office for National Statistics has published several papers about these revisions, and the 'quality' of its 'estimates' of GDP. In regard to the annual GDP estimates, Mahajan (2006) reports that looking at UK GDP at current prices for 1991 through to 2004, the change between the first estimate published and the latest available at the time of writing ranged from 0.4% to 2.8% of GDP.

19.3.1 The UNSTAT proposals: satellite accounting

The practical possibilities for environmental modifications to national income accounting conventions have been under active consideration by many individuals and institutions for a number of years. In the wake of the emergence of, and interest in, the idea of sustainable development, the United Nations Statistical Division (UNSTAT) has proposed draft guidelines for new national income accounting conventions, the System of Integrated Environmental and Economic Accounting (SEEA). Here we provide an informal outline of the essentials of the guidelines; a more formal account is given in Appendix 19.4.

The essential idea is to measure the 'environmental cost' of economic activity in a period. Environmental cost (EC) is defined as the difference between the opening and closing value of the stock of environmental assets

$$\mathrm{EC}_t \equiv \sum a_{it}v_{it} - \sum a_{it-1}v_{it-1} \tag{19.23}$$

where the summation is over $i = 1, 2, \ldots, n$ assets, a_i represents the physical measure of the ith environmental asset, v_i the unit value assigned to the ith asset, and where $t - 1$ refers to the start of the period and t to the end of the period. For the ith asset, $a_{it}v_{it} - a_{it-1}v_{it-1}$ is its depreciation over the period. EC_t is the change in the balance sheet value of all n environmental assets over the period, the depreciation of what is sometimes called 'natural capital'. In line with the discussion of the previous section, environmentally adjusted net domestic product could then be defined as

$$\mathrm{EDP}_t \equiv \mathrm{NDP}_t - \mathrm{EC}_t \equiv (\mathrm{GDP}_t - D_{\mathrm{M}t}) - D_{\mathrm{N}t} \tag{19.24}$$

where NDP stands for Net Domestic Product, D_{M} for the depreciation of human-made reproducible capital, and $D_{\mathrm{N}} \equiv \mathrm{EC}$ for the depreciation of natural capital.

The UNSTAT proposals do not envisage replacing the publication of the standard GDP/NDP accounts with the publication of EDP accounts. They do envisage complementing the standard accounts with balance sheets for natural capital, from which users of the accounts could work out EDP. This would leave intact the current conventions for the measurement of GDP and NDP, so that adoption of the proposal would mean that figures on these constructs would continue to be available on a consistent basis with past data. The balance sheets for environmental assets are, therefore, referred to as 'satellite accounts'. The potential, discussed below, for large year-on-year changes in estimates of the depreciation of non-renewable resources is another reason why most of those concerned with the production of national income accounts favour the satellite accounting approach, rather than producing only figures for environmentally adjusted national income. The idea is to publish each year conventional national income accounts accompanied by opening and closing balance sheet accounts for environmental assets.

In principle, the satellite accounts should cover all environmental assets relevant to production and consumption. This would require physical data and valuations for all relevant assets, and this is not now available even in those countries where the official statistical agencies have invested heavily in generating, collating and publishing environmental data. The problems are seen as especially acute with respect to valuation data for those assets not subject to market transactions. We shall see shortly, however, that even for mineral deposits subject to private property rights, there are quite serious problems about both physical data and valuation for depreciation. The UNSTAT proposals envisage that the range of assets used for the calculation of environmental cost be extended over time, starting with non-renewable resources and renewable resources involving market transactions.

As well as resource depletion and environmental degradation, we have noted that some commentators argue for the deduction of defensive environmental expenditures from the measure of NDP. The UNSTAT proposals do not involve treating defensive expenditures as an element of environmental cost for the adjustment of NDP to EDP, for two main reasons. First, as a practical matter, it is very difficult to definitively identify and measure such expenditures. Second, and more fundamentally, such subtraction might open the door to questioning the whole basis of measured national income as a welfare indicator. Leaving the natural environment aside, much of the expenditure counted in national income could be regarded as defensive – we eat and incur medical expenses to stay alive, we buy clothes to defend against the weather and social disapproval, and so on. The UNSTAT proposals do, however, involve identifying and separately reporting defensive environmental expenditures in the accounting system.

It must be emphasised that we have been discussing proposals and guidelines. As far as we can ascertain, no nation's official statistical office currently produces regular comprehensive satellite environmental accounts along with its standard national income accounts. A number have produced estimates of balance sheets for a limited set of natural resources, as exemplified by the satellite accounting for Australia and the UK discussed below in Section 19.3.3.[6]

19.3.2 Methods for measuring the depreciation of non-renewable resources

A somewhat extended treatment of this particular of the practice of environmental accounting is justified because, given that non-renewable resources are generally subject to private property rights and traded in markets, they are, from the general class of environmental assets, the case where it should be most straightforward to come up with numbers for depreciation. In fact, as we shall see, even in this case, obtaining a single 'correct' number for the depreciation of a particular resource is problematic.

As we have seen, the theoretically correct measure of the depreciation of an economy's stock of a non-renewable resource is the total Hotelling rent (THR) arising in its extraction. With P for the price of the extracted resource, c for the marginal cost of extraction, R for the amount extracted, N for new discoveries, and D for the depreciation of the resource stock:

$$D = \text{THR} = (P - c)(R - N) \tag{19.25}$$

If the standard assumptions for a fully competitive economy held, we would have

$$\text{THR} = \text{CIV}$$

[6] According to a recent World Bank publication (World Bank 2006, Table 9.1), 18 countries have environmental accounting programmes that regularly produce some environmental asset data. This source notes 'countless one-time or academic studies': for a listing of such go to UNSTAT's 'Searchable Archive of Publications on Environment-Economy Accounting' at http://unstats.un.org/unsd/envaccounting/ceea/archive/introduction.asp which can be searched by country.

where CIV is the change in the market value of the economy's stock of the non-renewable resource in question. In principle, and given the standard assumptions, D could be measured as either THR given by equation 19.25 or as CIV, with the same result.

In practice, neither of these measures of D appears to have been used, nor are they proposed for use in the literature concerning how environmental accounts might actually be constructed. The most obvious problem with equation 19.25 is that c, the marginal cost of extraction, is not observable in published, or readily available, data. As we shall see, there are other problems with using equation 19.25 to measure D. If there existed competitive firms that were solely in the business of selling the rights to extract from the resource stock, which they owned, then stock market valuations of such firms could be used to measure CIV. Generally such firms do not exist, resource ownership and extraction being vertically integrated in mining firms. Stock market valuations of mining firms are available, but these data confound the changes in other asset values with those of the mineral deposits owned, and reflect changes in overall stock market 'sentiment'. In any case, the minerals sector of an economy is rarely such that it can properly be characterised as 'competitive'.

There are four main methods that appear in the literature concerned with the practical implementation of environmental accounting.

19.3.2.1 Net price

This uses average cost, C, instead of marginal cost to compute rent, which is taken as the measure of depreciation, so that:

$$D = (P - C)(R - N) \tag{19.26}$$

Note that for $c > C$, $(P - C) > (P - c)$ so that on this account there would arise an overestimation of THR using equation 19.26. In some applications of the net price method, N is ignored. In what follows we shall refer to the net price method with new discoveries ignored as 'Net price I', and to the use of equation 19.26 with the N adjustment as 'Net price II'. Given that actual accounts refer to periods of time, rather than to instants of time as in the theoretical literature, applications of equation 19.26, with or without N, also vary as to the treatment of P and C in terms of dating. Clearly, each could be measured at the start or the end of the period, or as some average over the period. These three measures will only coincide if P and C are unchanging throughout the period, which in the case of P is uncommon.

19.3.2.2 Change in net present value

With 0 indicating the start of the accounting period and 1 its close, this method uses

$$D = \sum_{t=0}^{T_0} [(P_t - C_t)R_t/(1+i)^t] - \sum_{t=1}^{T_1} [(P_t - C_t)R_t/(1+i)^t] \tag{19.27}$$

where T_0 and T_1 are deposit lifetimes, and i is the interest rate. Apart from the use of C rather than c, this method can be seen as an alternative (to stock market valuations) method of measuring CIV. As actually used this method requires some specialising assumptions, as discussed below.

19.3.2.3 El Serafy's (user cost) rule

El Serafy was the economist who proposed this method (El Serafy, 1989), which is intended to measure depreciation as 'user cost'. User cost is the difference between net receipts actually arising and the constant perpetual income stream that is equivalent to the current value of the resource deposit. Given some special assumptions, this leads, as shown in Appendix 19.4, to the rule

$$D = R(P - C)/(1 + i)^T \tag{19.28}$$

where r is the interest rate, and T is the deposit lifetime assuming a constant rate of extraction.

19.3.2.4 Comparing results from four methods

It is generally understood that the net price method is liable to produce large year-on-year fluctuations in estimated D, and this method is not recommended in the UNSTAT guidelines for environmental accounting. Those guidelines recommend the change in net present value method. The net price method is, however, quite widely used by analysts seeking a figure for D in respect of non-renewable resource stocks. It avoids the need for information/assumptions about mine lifetimes and interest rates, which arises with

Table 19.3 Alternative estimates of minerals depreciation for Australia 1988/9 to 1991/2, AUS$ $\times 10^6$

Year	El Serafy rule	Net price I	Net price II	ABS NPV change
1988/89	952	8 511		
1989/90	1228	9 872	−19 321	−6 500
1990/91	1922	12 023	−147 035	−19 900
1991/92	2328	13 624	299 075	−9 700

the other methods. It is instructive to consider the application of different methods to a common body of data.

We can do this because in 1995 the Australian Bureau of Statistics (ABS) produced detailed preliminary balance sheet estimates for a range of assets including 'Subsoil assets', that is, non-renewable mineral resources (ABS, 1995).[7] Table 3.3 of ABS (1995) gives for each of 33 minerals:

1. the size of 'Economic Demonstrated Resources' (EDR) at 30 June 1989, 1990, 1991 and 1992;
2. the price of the extracted mineral at 30 June 1989, 1990, 1991 and 1992;
3. the average cost of extraction at 30 June 1989, 1990, 1991 and 1992;
4. production of the mineral in the years ending 30 June 1989, 1990, 1991 and 1992.

From these data the ABS calculated for each of the 33 minerals, for each of four years, the NPV of the resource stock according to

$$\mathrm{PV}_s = \sum_{t=1}^{T_s} [(P_t - C_t)R_t/(1+i)^t]$$

where s refers to the balance sheet date (30 June 1989, 1990, 1991 and 1992) and T_S is the estimated stock lifetime at that date. Several specialising assumptions are made. First, R_t is set equal to R for all t where:

$$R = 0.25(R_{88/9} + R_{89/90} + R_{90/1} + R_{91/2})$$

At each s the resource lifetime, T, is calculated as:

$$T_s = \mathrm{EDR}_s/R$$

It is assumed that for all t, $(P_t - C_t) = (P_s - C_s)$. Using an interest rate of 7.5% the PV_s results were summed across minerals to give balance sheet valuations of Australia's 'subsoil assets' at 30 June 1989, 1990, 1991 and 1992. From year-on-year changes in the balance sheet figures, we can obtain estimates of the depreciation of the minerals stock, shown in the fifth column of Table 19.3, under 'ABS NPV change'.[8]

The figures shown in Table 19.3 under 'El Serafy rule' are calculated according to equation 19.28 using the same P, C and T data from ABS (1995), and the same 7.5% interest rate. The 'Net price II' figures use the net price method, taking account of new discoveries, calculating rent as $(P_s - C_s)(R_s - N_s)$ where P_s, C_s and R_s are taken direct from the ABS data, and N_s is inferred from $(\mathrm{EDR}_s - \mathrm{EDR}_{s-1})$ and R_s in those data. ABS (1995) notes a number of problems regarding the EDR data. Of particular concern are changes due to variation in P, technological change, and revisions of resource classification. The figures for 'Net price I' in Table 19.3 are the least affected by specialising assumptions and the problems attending the EDR data, as they ignore new discoveries, being calculated as $(P_s - C_s)R_s$.

From a comparison of equation 19.26, with N equal to zero, and equation 19.28, it is clear that, for the same data, the 'Net price I' figure must always be larger than the 'El Serafy rule' figure. From equation 19.26, assuming that N must be non-negative means that 'Net price II' must always be smaller than 'Net price I'. In Table 19.3, for 1989/90 and 1990/1 across all minerals new discoveries are positive and large enough to produce a negative figure for depreciation. However, for 1991/2 'Net price II' depreciation is much larger than 'Net price I' depreciation. This is primarily because of a reduction in the EDR figure for bauxite for 30 June 1992 by an amount which exceeds the previous year's production, 0.04 gigatonnes, by 3.96 gigatonnes, implying negative new discoveries. The pattern in the 'ABS NPV change' figures follows that for 'Net price II' with dampened swings, as would be expected given the method of calculation described above.

[7] For further information on the ABS data and details of the calculations for Table 19.3 here, see Common and Sanyal (1998).

[8] The ABS also did the calculations for interest rates of 5% and 10%. Their results illustrate the sensitivity of depreciation as measured by the change in NPV method to the interest rate used.

Which of these is the 'correct' way to measure depreciation? The capital-theoretic approach could be taken to suggest 'Net price II', were it not for the fact that it uses average, rather than marginal, costs. Given marginal costs greater than average costs, one could argue, this measure should be taken as an upper bound on depreciation. However, given the problems about measuring 'new discoveries', the implications of this argument in practice can, as shown in Table 19.3, lead to negative figures for non-renewable-resource depletion. The change in NPV method, as implemented by ABS, also gives rise to negative depreciation for minerals, as shown in the final column of Table 19.3. A negative figure for depreciation means that the effect of accounting for the depletion of non-renewable resources would be, other things equal, to make sustainable national income larger than conventionally measured national income. Notice also that the 'Net price II' and 'ABS NPV change' results in Table 19.3 are more volatile than those for 'El Serafy rule' and 'Net price I'. Using either of the former to adjust conventionally measured national income would make the time series resulting highly volatile.

The point of reporting these results is not to criticise the work of the Australian Bureau of Statistics. It is to demonstrate that different methods for going from a common set of physical data on stocks and flows of non-renewable resources to monetary estimates of depreciation lead to substantially different numbers for those depreciation estimates. The same point was demonstrated in two papers, Bryant and Cook (1992) and Vaze (1998a), from the UK's official economic statistics agency giving early estimates for the depreciation of UK oil and gas reserves. We now look at more recent data from the UK and Australia on the value of non-renewable natural resources.

19.3.3 Satellite accounts for the UK and Australia

In the UK, the official statistics agency now regularly publishes figures for the value of the country's oil and gas reserves, from which depreciation figures can be inferred. These are the only environmental asset monetary estimates for the UK available from official sources. These oil and gas satellite accounts appear in the main annual national income accounts publication, known as '*The Blue Book*' – see, for example, Office for National Statistics (2008b). This also gives the corresponding physical data for oil and gas reserves, as well as data, physical and monetary, on other matters environmental. All of these physical and monetary data also appear, along with some other data, in the agency's bi-annual environmental accounts publication – see for, example, Office for National Statistics (2008a).[9]

Table 19.4 shows the values given for the UK's oil and gas reserves in *The Blue Book* for 2008. These were calculated on the present value method, and as the source notes the figures are 'extremely

Table 19.4 UK asset values 1999–2007

	£ billion end year				
	Oil	Gas	Oil + Gas	Non-financial Assets	Residential Buildings
1999	46.964	30.495	77.459	3877.5	1848.9
2000	53.611	43.011	96.622	4245.1	2106.5
2001	51.812	50.451	102.263	4484.8	2267.8
2002	50.883	46.566	97.449	5076.8	2737.1
2003	53.045	44.250	97.295	5522.2	3054.9
2004	78.536	50.754	129.29	6069.0	3427.0
2005	100.192	65.402	165.594	6283.0	3555.0
2006	120.921	69.439	190.36	6863.1	3915.3
2007	177.891	68.340	246.231	7380.0	4313.6

Source: Office for National Statistics 2008a

[9] Both of these publications can be accessed at websites given in Further Reading at the end of the chapter.

sensitive to the estimated return to capital (i.e. the interest rate) and to assumptions about future unit resource rents' (parenthesis added). Table 19.4 also shows the value given there for all of the UK's non-financial assets – buildings, equipment, infrastructure, vehicles etc. The final column of Table 19.4 shows the value of the UK's housing stock, which in 1999 accounted for 47.7% of the total non-financial asset value, and in 2007 accounted for 58.5%. Whereas the value of assets other than residential buildings grew by 51% over 1999 to 2007, the value of the housing stock increased by 133%. The latter was mainly down to the widely noted rise in house prices rather than an increase in the size of the housing stock. In 1999 oil and gas reserves were valued at 2.0% of the total value of non-financial assets, and in 2007 the ratio was 3.3%. If we take the housing stock out of the total, the proportion of it accounted for by oil and gas was 4.4% in 1999 and 8.0% in 2007. The rise in the value of oil and gas reserves over this period was due to increases in their prices – the physical data which accompany the asset value data in the source for those data show that both oil and gas reserves were lower in 2007 than in 1999.

By taking the differences between end year asset values one can calculate the depreciation of the UK's oil and gas reserves for the corresponding year. The Office for National Statistics does not publish the results of such calculations. They are plotted in Figure 19.2 here. A negative figure means that the estimated value of the reserve increased over the year, which could be on account of new discoveries exceeding extraction and/or an increase in unit rent. In Figure 19.2, of the eight years, six show negative depreciation for oil, five for gas, and six for the total. Examination of the physical data shows that for oil physical depletion was positive in five of the eight years, as it was for gas in seven of the eight. The predominance of annual negative depreciations shown in Figure 19.2 was mainly on account of unit rent changes. In 2008 world oil and gas prices started to fall.

The official Australian statistical agency, ABS, now produces satellite accounts with a wider environmental coverage than its UK counterpart. Table 19.5 uses data from the *Year Book Australia 2008*

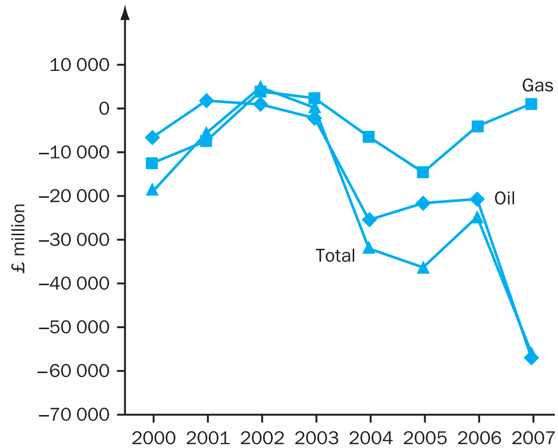


Figure 19.2 Oil and gas depreciation for the UK, 2000–2007

Table 19.5 Australian asset values 2002–2006

	$billion 30th June				
	2002	2003	2004	2005	2006
Total NFA	4004	4435.9	5014.8	5391.4	5876.7
Produced	2150.0	2291.5	2482.5	2702.1	2932.9
Machinery and equipment	346.9	352.3	361.2	382.6	409.3
Dwellings	812.4	892.5	991.6	1086.2	1172.1
Non-produced	1854.7	2144.3	2532.3	2689.3	2943.8
Land	1639.8	1920.4	2284.0	2417.7	2633.3
Subsoil	204.9	213.6	237.2	260.2	298.8
Forest	1.9	2.0	2.1	2.2	2.2

Source: ABS 2008, National Balance Sheet, Table 30.17

(ABS 2008).[10] NFA stands for non-financial assets. The ABS distinguishes between Produced and Non-produced assets, with the reported environmental assets comprising approximately 99% of the latter. As shown in Table 19.5, those environmental assets are land, subsoil and forests. Subsoil assets include all economically significant non-renewable energy and mineral resources, valued using the present value method. Forests are native forests, with plantation forests being counted as produced assets. The value of the latter is approximately four times that of the former. Both kinds of forest are valued using the commercial value of the standing timber. Australia is a country, where by the standards of rich economies,

[10] The data were downloaded from the electronic version of the yearbook at http://www.abs.gov.au/ausstats/abs@nsf/mf/1301.0.

the mining sector is very important. The estimated value of subsoil assets is approximately 5% of the total value of non-financial assets, less than that of machinery and equipment, about a quarter of that of dwellings. In value terms, land dominates the non-produced asset classification.

Even in the Australian case, the coverage of environmental assets in the satellite accounts is limited. Apart from native forest, there are no estimates relating to renewable resources. There are none for water resources of any kind. Nor are there any relating to the amenity services that the natural environment provides. Australia's natural environment is important to its tourist industry, which is now widely considered to be as important to its economy as mining or agriculture.

19.3.4 Environmentally adjusted national income results

As noted above, the UNSTAT proposals do not envisage replacing the publication of the standard GDP/NDP accounts with the publication of EDP accounts, and as far as we can ascertain no official statistical agency has done this.[11] As we have just seen, in some countries the agency does regularly publish satellite accounts for some environmental assets. In this section we look at three exercises in using such data to produce a number for environmentally adjusted NDP, which we have given the acronym EDP. Only in the third case are the numbers for EDP produced by an official national statistical agency.

19.3.4.1 Indonesia

We look first at what appears to have been the first attempt to produce numbers for EDP. The quotation from Repetto *et al.* (1989) which heads this chapter is from the introduction to a report in which a World Resources Institute (WRI) team adjusted official national income measures for Indonesia by WRI estimates of the depreciation of three environmental assets considered important to that country – oil deposits, timber and soil. WRI proceeded by first constructing physical accounts, then applying unit values. In the case of oil opening stocks were valued at the current market price of the extracted oil less estimated average extraction cost. Closing stocks were computed by subtracting extraction during the year and adding new discoveries, and valued in the same way as the opening stocks using the price ruling at the end of the period. This is an application of the Net price II method. The procedure followed with timber is the same except that it allows for estimated natural growth over the year. The physical data here were recognised as being less firmly based than in the case of oil. For soil erosion, estimated physical losses over the year were valued using estimates of the loss of agricultural output entailed.

Table 19.6 reports, in the second and third columns, the GDP and EDP estimates from Repetto *et al*. in index number form, where EDP is GDP minus the depreciation of the three environmental assets considered. The average per annum growth rates are 7.1% for GDP and 4.1% for EDP. EDP grows more slowly than GDP over the period 1971–1984, and behaves more erratically. The fourth column shows the ratio of the EDP estimate to GDP. The erratic

Table 19.6 GDP and an EDP estimate for Indonesia 1971–1984

Year	GDP Index	EDP Index	EDP/GDP
1971	1	1	1.20
1972	1.09	0.90	0.99
1973	1.22	0.97	0.96
1974	1.32	1.48	1.36
1975	1.38	0.98	0.85
1976	1.47	1.12	0.92
1977	1.60	1.08	0.81
1978	1.73	1.19	0.78
1979	1.83	1.19	0.78
1980	2.01	1.28	0.76
1981	2.17	1.48	0.82
1982	2.22	1.58	0.86
1983	2.32	1.49	0.78
1984	2.44	1.68	0.83

Source: Based on Repetto *et al.* (1989)

[11] In chapter 9 of World Bank (2006) it is stated that 'a number of countries have calculated partially adjusted eaNDPs, including Germany, Japan, the Republic of Korea, the Philippines, and Sweden' (eaNDP = EDP here). Unfortunately no references are given in support of this statement. When we visited the websites for these countries' statistical agencies we were unable to find any reference to any environmental adjustments to national income measures. Three pilot studies (for Mexico, Papua New Guinea and Korea) conducted in association with UNSTAT can be accessed at http://unstats.un.org/unsd/envaccounting/cprojects.asp.

Table 19.7 UK GDP, NDP and NDP adjusted for oil and gas depreciation

	2001	2002	2003	2004	2005	2006	2007
GDP	1 021 828	1 075 564	1 139 746	1 200 595	1 252 505	1 321 860	1 401 042
−FCC	115 796	121 914	125 603	135 184	138 520	147 858	158 143
=NDP	906 032	953 650	1 014 143	1 065 411	1 113 985	1 174 002	1 242 899
−DEPCTN	−5 641	4 814	154	−31 995	−36 304	−24 766	−55 871
=EDP	911 673	948 836	1 013 989	1 097 406	1 150 289	1 198 768	1 298 770
GDP growth		5.3%	6.0%	5.3%	4.3%	5.5%	6.0%
NDP growth		5.3%	6.3%	5.1%	4.6%	5.4%	5.9%
EDP growth		4.1%	6.9%	8.2%	4.8%	4.2%	8.3%

Source: based on data from Office of National Statistics 2008b

behaviour of the EDP series is principally due to the effect of changes in the price of extracted oil, and of new discoveries of oil. The EDP figures for 1973 and 1974 show the effects of the increase in the world price of oil. If EDP is understood as sustainable income, these figures show sustainable income increasing by 51% in one year, 1973 to 1974.

19.3.4.2 The UK

As seen above, the UK's national statistical agency does now produce satellite environmental accounts, albeit for just oil and natural gas. It does not use the estimates in these accounts to produce any adjusted national income figures. We have done this in Table 19.7.

The figures in the upper part of the table are in units of millions of £s. All figures are current values, i.e. they are not adjusted for inflation. FCC stands for Fixed Capital Consumption, which is the depreciation of reproducible capital used in the standard adjustment from Gross to Net Domestic Product. The figures shown against DEPCTN in Table 19.7 are the end year to end year changes in the balance sheet figures for Oil+Gas in Table 19.4. An increase in Table 19.4 is shown as a negative entry in Table 19.7 so as to preserve the convention of subtracting depreciation. These figures are taken straight from *The Blue Book*. We use them to calculate the figures shown for EDP in Table 19.7. Given negative depreciation in some years, EDP is sometimes larger than NDP. Note that oil and gas depreciation varies a lot more than FCC and is usually a lot smaller. In the lower part of Table 19.7 we show the growth rates for GDP, NDP and EDP.

19.3.4.3 Australia

In Australia the ABS does not use its satellite environmental accounts estimates to adjust the national income figures that it reports in its main publications in that area. It did, however, do that in the section of *Year Book Australia 2008* dealing with 'Environmental Assets'. Australia may well be the only country where the national statistical agency responsible for national income accounting has published numbers for environmentally adjusted NDP.

Figures from Table 2.20 of the Year Book are reproduced in the upper part of Table 19.8: units are millions of current value $s (there is no adjustment for inflation). FCC has the same meaning as in Table 19.7. ADJSTMNT is what ABS calls the 'net depletion adjustment', which is subsoil (i.e. fossil fuels and metallic minerals) depletion (extraction), plus land degradation, less subsoil additions.[12] The net depletion adjustment is always a positive number so that what we call EDP, and ABS calls 'Depletion adjusted NDP', is always less than NDP. In the last two years subsoil depletion is smaller than subsoil additions, and the net depletion adjustment is positive because land degradation is larger in absolute value than the negative subsoil net depletion.

The lower part of Table 19.8 shows growth rates: for GDP, NDP and EDP they are reproduced from the ABS Table 2.21, for GDP pc (per capita) they are calculated from the figures for GDP in the first

[12] The figures for subsoil depletion and subsoil additions given in Table 2.20 of the yearbook imply net changes which are substantially different from those implied by the figures for subsoil asset values given in Table 30.17 of the yearbook and reproduced in Table 19.5 here.

Table 19.8 Australian GDP, NDP and NDP after net depletion adjustment

	2001/2	2002/3	2003/4	2004/5	2005/6
GDP	735 714	781 675	840 285	896 568	965 969
−FCC	115 259	121 526	127 754	134 523	145 476
=NDP	620 455	660 149	712 531	762 045	820 493
−ADJSTMNT	1 317	865	894	87	234
=EDP	619 138	659 284	711 637	761 958	820 259
Growth rates					
GDP	6.7%	6.2%	7.5%	6.7%	7.7%
NDP	6.6%	6.4%	7.9%	6.9%	7.7%
EDP	6.5%	6.5%	7.9%	7.1%	7.7%
GDP pc		5.0%	5.6%	6.0%	6.3%

Source: ABS (2008), Environmental Assets, Tables 2.20, 2.21 and 7.1

row of Table 19.8 and population data from elsewhere in the Year Book. On these figures growth rates for NDP and EDP are not much different. Australia is a country where, for an affluent developed nation, the population growth rate is high. Adjusting GDP for population growth has a bigger effect on the growth rate than adjusting for FCC and net depletion. Although it is widely agreed that interpretation as a measure of economic performance requires that GDP, or NDP, is adjusted for population growth, commentary usually focuses on the unadjusted figure.

19.4 Wealth and genuine saving

In this section of the chapter we bring together two recent developments in environmental accounting. The first is largely a matter of emphasis and presentation, whereby attention is focused on properly measuring wealth and total saving, which is the same as total investment, rather than income. An inclusive measure of total saving/investment is now widely referred to as 'genuine saving'. The second is a complete departure from what we have already looked at, in that it seeks to provide a theoretical basis for environmental accounting that is not restricted to an optimising economy. The link between the two developments is that the theory developed for a non-optimising economy, an 'imperfect economy', points to the use of genuine saving/investment for monitoring performance with respect to sustainable development.

19.4.1 Genuine saving

We begin with a restatement of the basic ideas as first set out in Pearce and Atkinson (1993): see also Atkinson and Pearce (1993) and Pearce and Atkinson (1995).

Following on from our discussion of the proper measurement of national income, with I_{Rt} for net investment in reproducible capital and D_{Nt} for the depreciation of natural capital we can write

$$\text{EDP}_t = C_t + I_{Rt} + D_{Nt} \tag{19.29}$$

Sustainable income is often defined as the maximum that can be consumed without running down the stock of capital. By equation 19.29

$$\text{EDP}_t > C_t \text{ for } (I_{Rt} + D_{Nt}) > 0$$

$$\text{EDP}_t = C_t \text{ for } (I_{Rt} + D_{Nt}) = 0$$

$$\text{EDP}_t < C_t \text{ for } (I_{Rt} + D_{Nt}) < 0$$

so that maximum consumption consistent with not running down the capital stock is $C_t = \text{EDP}_t$, so that EDP_t is sustainable income. Then we can say that sustainable development requires:

$$C_t \leq \text{EDP}_t \tag{19.30}$$

Note that $C_t = \text{EDP}_t$ implies that I_{Rt} and D_{Nt} are equal and of opposite sign so that $(I_{Rt} + D_{Nt}) = 0$.

With K_{Rt} for reproducible capital and K_{Nt} for natural capital we can write

$$W_t = K_{Rt} + K_{Nt} \tag{19.31}$$

where W stands for wealth as the aggregate capital stock. For W_{t+1} we can write

$$W_{t+1} = (K_{Rt} + I_{Rt}) + (K_{Nt} + D_{Nt})$$

so that

$$W_{t+1} - W_t = I_{Rt} + D_{Nt}$$

which by equation 19.29 is

$$W_{t+1} - W_t = \text{EDP}_t - C_t \tag{19.32}$$

so that $W_{t+1} - W_t \geq 0$ if $C_t \leq \text{EDP}_t$.

Hence,

$$W_{t+1} - W_t \geq 0 \tag{19.33}$$

is equivalent to the expression 19.30 as a test for sustainable development. $W_{t+1} - W_t$ is what is now widely known as 'genuine saving' or 'genuine investment' for period t. The argument is that if genuine saving is non-negative, then a necessary condition for sustainable development is being met.

We arrived at the expression 19.33 considering just two kinds of capital, reproducible and natural. The argument would work in exactly the same way if we had started with

$$\mathrm{EDP}_t = C_t + I_{\mathrm{R}t} + D_{\mathrm{N}t} + I_{\mathrm{S}t} + I_{\mathrm{H}t}$$

instead of equation 19.29, and used

$$W_t = K_{\mathrm{R}t} + K_{\mathrm{N}t} + K_{\mathrm{S}t} + K_{\mathrm{H}t} \tag{19.34}$$

instead of equation 19.31, where the subscripts S and H denote social and human capital/investment respectively. In so far as social and human capital are inputs to economic activity, equation 19.34 is the proper definition of wealth as total asset value, and a measure of genuine saving should include (net) investment in those kinds of capital. It is now widely understood that social and human capital are, in fact, very important inputs to economic activity.

19.4.2 Theory for an imperfect economy

The theory that supports EDP as the proper measure of national income, looked at in section 19.2 above and Appendices 19.1 and 19.2, uses results from optimisation models. As actual economies are definitely not optimised, this troubles some economists. It is not obvious that results derived for optimising economies can be transferred to, and used to guide policy in, actual economies. Some economists have, therefore, sought to derive results that do not require the assumption of optimisation and which can be applied to what they call 'imperfect', i.e. actual, economies. The main result is that non-negative genuine saving/investment at time t goes with sustainable development at time t. In setting out the basic ideas here we draw mainly on Dasgupta (2001): other references appear in Further Reading at the end of the chapter.

It is assumed here for simplicity that the size of the population is constant. Social well-being at time t is taken to be the discounted sum of current and future utility

$$V_t = \int_{\tau=t}^{\infty} U(C_t)\mathrm{e}^{-\rho(\tau-t)}\mathrm{d}t \tag{19.35}$$

where C an aggregate index of the determinants of U, i.e. the things that people care about such as produced commodities and environmental amenities. A consumption stream beginning at $t = 0$ is said to correspond to a sustainable development path at t if $\mathrm{d}V_t/\mathrm{d}t$. Note that this idea of what constitutes sustainable development is defined only for a point in time. It requires, however, that in terms of the discounted sum of all future utility the state of the economy is non-deteriorating.

It can be shown, see Appendix 19.3, that $V_{t+1} \geq V_t$ is equivalent to

$$I_t^G = \sum_{i=1}^{N} p_{it}\frac{\mathrm{d}A_{it}}{\mathrm{d}t} \geq 0 \tag{19.36}$$

where I_t^G stands for genuine investment (equal to genuine saving), $\mathrm{d}A_{it}/\mathrm{d}t$ is the change in the stock of the ith asset that affects U directly or indirectly (via its role in production), and p_{it} is the accounting price for the ith asset. If the expression 19.36 holds, that is, the economy is at time t developing sustainably. In words, if relevant asset size changes are aggregated using accounting prices, non-negative genuine saving/investment tells us that development has been sustainable in the sense defined above. And, for this to be the case we do not need to assume that the economy in question is an optimising economy – it applies to any actual economy.

This remarkable result depends crucially on what is meant by an accounting price, p_{it}. The accounting price for asset i at time t is the change in V_t consequent upon an infinitesimally small change in the amount of the asset at time t, other things remaining the same. In order to know an asset's accounting price a would-be accountant would have to be able to figure out what effect a small change in its size would have on current and future utility. This is a large task. As Dasgupta puts it:

> accounting prices depend upon four related factors: (a) the conception of social well-being, (b) the size and composition of existing stocks of assets, (c) production and substitution possibilities among goods and services, and (d) the way resources are allocated in the economy.
>
> *(Dasgupta 2001 p. 123)*

The price of getting away from results based on the assumption of optimisation is the assumption that the accountant can forecast all of the utility consequences of small perturbations in all relevant asset stock sizes through to the distant future.

19.4.3 Problems with genuine saving as a sustainability test

Clearly, no accountant for an actual economy could have the information necessary to compute and use all of the accounting prices relevant to that economy to properly compute genuine saving, as that information comprises complete knowledge of current changes in stocks of all assets and complete knowledge of the future. What is actually envisaged for genuine saving measurement is the use of a wider range of assets than those which comprise reproducible capital, and for those 'extra' assets to be valued using estimates of accounting prices based on partial models of the economy. The implicit claim made by proponents of genuine saving as a sustainability test is that aggregating over a wider range of assets using estimates of accounting prices will result in a better guide to policy than looking just at investment in reproducible capital.

This claim is examined in Common (2007b), where it is found that it is not supportable as a general proposition. Table 19.9 here reproduces a constructed numerical example from that paper which shows that it is possible for an extended but incomplete accounting to be further from the truth than just looking at reproducible capital. The example distinguishes four classes of asset, and uses the notation introduced at equation 19.34 above.

Table 19.9 Numerical example for incomplete genuine saving accounting

Time	K_R	K_N^I	K_S^I	K_H^I	K_N^O	K_S^O	K_H^O	W
0	100	1000	100	100	500	100	100	2000
1	102	950	101	101	550	110	120	2034
Change	2	−50	1	1	50	10	20	34
	K_R	K_N^e	K_S^e	K_H^e				W^e
0	100	1100	50	50				1300
1	102	1000	51	51				1204
Change	2	−100	1	1				−96

Source: Common (2007b)

For simplicity only, it is assumed that all assets comprising reproducible capital are accurately accounted for. In regard to the other asset classes the example assumes that not all assets are accounted for in the extended measure of genuine saving, and that for those assets that are included, what gets included is an estimate. Thus, for example, K_N^I is the true value of the aggregate of the natural capital assets that get included in the extended measure, K_N^O is the true aggregate value of those left out, and K_N^e is the estimated value that gets used in the extended measure. K_N^I and K_N^e differ on account of incorrect estimation of quantities and/or prices. Time 0 and 1 refer to the start and end of the accounting period.

As shown in the top part of Table 19.10, genuine saving is actually 34, whereas just looking at reproducible capital would indicate saving/investment of 2. The bottom part of the table shows measured genuine saving as −96, which is of the opposite sign to actual genuine saving, and entails a larger error than just looking at the accumulation of reproducible capital.

The point here is not that extended but incomplete and inaccurate measurement of genuine saving always produces more error than measuring just reproducible capital accumulation, but that it may do. As shown in Common (2007b), it is possible to derive a statement of the conditions under which we would know that the extended measure of genuine saving involved a smaller absolute error than the conventional measure as investment in reproducible capital. Unfortunately that statement includes terms for the changes in K_N^O, K_S^O, and K_H^O, which are not observed, so that the statement cannot actually be used to decide the matter. The situation would be the same if it were the case that all relevant assets were accounted for, but using estimates rather than the correct values. If it were the case that the correct values were used for an incomplete set of assets, then we could say in any particular case whether including some assets other than reproducible capital assets made the measurement of genuine saving more or less accurate.

19.4.4 World Bank estimates

Pearce and Atkinson (1995) reported results for genuine saving for 18 countries, for 8 of which they have it as negative. All of these 8 are developing countries. They consider only natural capital. The exact asset coverage for D_N by country was not reported. Neither was the method by which asset depreciation was estimated. It appears that both coverage and method varied across countries.

The idea of genuine saving was taken up by the World Bank, which in recent years has put quite a lot of effort into comprehensive wealth accounting. Here we report and discuss World Bank methods and estimates as presented in a recent publication (World Bank 2006): a fuller account of the methods and data sources used by the World Bank to estimate genuine saving is given in Bolt *et al.* (2002). What we have been calling genuine saving the World Bank sometimes, within the same publication, refers to as genuine saving and sometimes as adjusted net saving, or savings. This can be confusing. We will stick with genuine saving.

The World Bank estimates are calculated as

Genuine saving = Gross saving
– Depreciation of fixed capital
+ Educational expenses
– Depletion of natural resources
– Pollution damages

where

Gross saving is Gross National Income (GNI) less the sum of private and public sector consumption, plus net foreign transfers.

Depreciation of fixed capital is the depreciation of what we have been calling reproducible capital, and is the replacement value of the capital used up in the process of production.

Educational expenses are measured as public current operating expenditures. Private sector spending on education is not included. Nor is public sector capital expenditure.

Depletion of natural resources is measured as the sum of energy (coal, gas and oil), minerals and net forest depletion. For energy and minerals, depletion is measured as rent equal to production multiplied by price minus average production cost. This is the Net price I method as discussed above. The same method is used for net forest depletion save that price minus cost is multiplied by the excess of harvest over natural growth.

Pollution damages are CO_2 damages, which, for every country, are measured as its emissions as tonnes of carbon multiplied by \$20, which is taken to be a conservative estimate of marginal global damages per tonne of carbon.[13]

The following comments arise:

a. For the data series used there are many missing observations, where the gaps have been filled in by interpolations of various kinds.
b. The only way in which changes in human and social capital stocks are accounted for is by public sector current expenditure. As a measure of changes in human and social capital stocks this is clearly inadequate. As a measure of educational investment it is quite arbitrary.
c. As regards the resource depletion side of natural capital, there are very many biotic assets which are not accounted for. Aquifers are a very important asset in many countries. The use of the Net price I method is not generally regarded as appropriate.
d. As regards negatively valued natural capital assets, only CO_2 is considered. It is only one of, albeit the most important of, the greenhouse gases. The use of country *x*'s emissions as the basis for calculating climate change damage is clearly inappropriate – a key feature of the climate change problem is that the damage to country *x* will bear no relationship to its current, or indeed its cumulative, emissions.

Overall, one might wonder whether these estimates should be taken at all seriously. The World Bank clearly intends that they should be. At the least, the publication of such estimates should be accompanied by a strong prominent statement as to their limitations, particularly as a guide to policy decisions. The World Bank does spell out reasonably

[13] This refers to the genuine saving estimates that are reported here in Figures 19.3 and 19.4 which were taken from the World Bank website in February 2009: go to http://www.worldbank.org and search on 'Adjusted Net Savings'. Those reported in the book (World Bank, 2006) also include a figure for the damage to human health done by particulate matter, which data are not available on the website (as of 15 March 2009).

clearly how its estimates are put together, and does offer some caveats, but they are neither strong nor prominent. In a publication directed at a non-specialist readership (World Bank, 2006) the Executive Summary says nothing about the incomplete basis for the estimates it discusses, or about the implications arising. The most explicit caveat offered (p. 38) is that 'we should be cautious in interpreting a positive genuine saving rate' as 'There are some important assets omitted from the analysis . . .'. From the discussion of the preceding sub-section it is clear that we should also be cautious in interpreting a negative genuine saving rate estimate.

Figures 19.3 and 19.4 reproduce World Bank genuine saving estimates, as percentages of GNI, for groups of nations. In Figure 19.3 the grouping is according to income level, and the estimates are for 1970 to 2004. For what it is worth, these results suggest that low income nations as a group have improved their genuine saving performance over this period. Figure 19.4 covers the period 1974 to 2004. According to these results, for the world as a whole, genuine saving was positive and fairly constant at around 10% of world GNI throughout this period. The estimates for the Middle East and Africa are strongly influenced by oil and gas extraction, and swings in the world prices for those things. These estimates are consistent with countries in this group spending some of the rents arising in such extraction on current consumption rather than investment. East Asia is generally regarded as a region where gross saving rates are high. Figure 19.4 suggests that, with the exception of the late 70s and early 80s, natural capital depletion was not sufficient to push the genuine saving rate much below that for the world as a whole. Most of the oil and gas from the Middle East and Africa is exported, to among other places East Asia, where there is not a lot of extraction of oil and gas. This raises a question which we explore in the next sub-section.

Before looking at how the fact of international trade can be handled, it is worth looking briefly at another part of the World Bank's work on the wealth-based approach to environmental accounting. In the publication where the genuine saving estimates are reported (World Bank, 2006) there are also reported estimates of total wealth and its components. Total wealth is estimated as the present value of future consumption, and 'Intangible capital', which is the sum of K_H and K_S here, is what is left of total wealth after subtracting Natural capital and 'Produced capital'. Given the way in which total capital is estimated, and the fact that Intangible capital is a residual, these estimates are not very interesting or informative.[14] However, the relativities

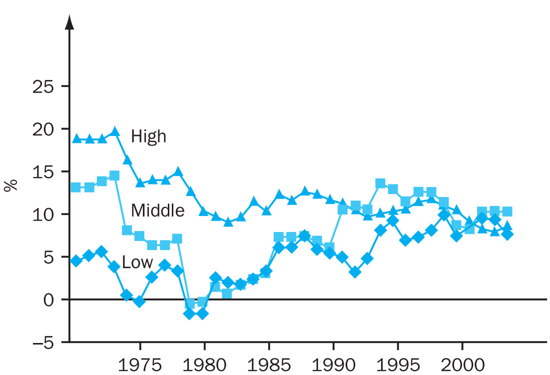


Figure 19.3 Genuine saving by income group

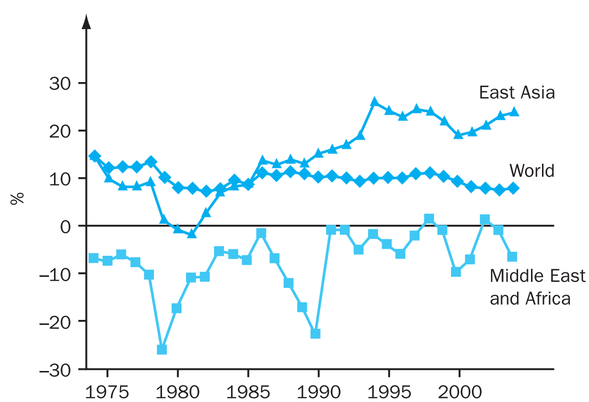


Figure 19.4 Genuine saving for selected regions and the world

[14] Wealth can be shown to be discounted future consumption only under restrictive assumptions. In the World Bank calculations, (World Bank, 2006), average annual consumption levels for 1998–2000 were projected forward at an unspecified growth rate. Where genuine saving was negative, it was subtracted from consumption to get sustainable consumption as the entry to the averaging. The same discount rate was applied to future consumption levels for all countries. Apart from the adjustment to get sustainable consumption, it appears that for all countries wealth is the same transformation on current average consumption, and so adds no information to that data.

in regard to the relative sizes of produced and natural capital, and the composition of natural capital, are of some interest.

For the wealth-level calculations, Produced capital comprises Machinery, equipment and structures and Urban Land. Natural capital comprises: Energy and mineral resources, Timber resources, Nontimber forest resources, Cropland, Pastureland, and Protected areas. Why the list of natural assets is different for estimating wealth and changes in wealth is not explained, but may be, at least in part, on account of data availability. For Energy and minerals, and Timber, asset values were obtained by the Net Present Value method. Again, the difference in method as between asset value estimation and change in asset value estimation is not explained. The remaining asset values were also arrived at using the Net Present Value method. All of the natural capital asset values are conditional on the discount rate used. Details of the methods for the figures that go into the NPV calculations are given in the source (World Bank, 2006). Protected area values per hectare are taken to be the opportunity cost of protection, the lower of the annual return per hectare for cropland and pastureland.

The first striking feature of Table 19.10 confirms expectations – per capita asset values, for all assets, increase with per capita income. The second is that the ratio of the value of produced capital to that of natural capital increases with income – 0.61 for low, 1.53 for middle, and 7.99 for high income. The share of natural capital accounted for by agricultural land decreases with income – 69% for low, 57% for middle, and 37% for high income. The share of Subsoil assets (energy and minerals) in natural capital increases with income – 17% for low, 31% for middle, and 40% for high income.

19.4.5 Accounting for international trade

As noted in Pearce and Atkinson (1995), given international trade, one nation's inhabitants can depreciate natural capital in another nation. Thus, for example, Japan is frequently cited as a country which has high domestic saving and investment in man-made capital, and low natural capital depreciation domestically, but which is responsible for much natural capital depreciation overseas when it imports raw materials. The obverse case would be somewhere like Saudi Arabia, where natural capital depreciation is high on account of exported natural resource extraction. Proops and Atkinson (1996) proposed a method which modifies the measurement of resource depreciation so as to allow for trade effects. The essential idea is to treat each economy as a sector in the global economy, and to use the techniques of input–output modelling discussed in Chapter 8.

Consider two trading economies, 1 and 2. Let x_{12} be exports from 1 to 2, and x_{21} be exports from 2 to 1. Let y represent total output, and f represent final demand, comprising c for consumption and s for saving/investment. We can then write:

$$\begin{aligned} y_1 &= x_{12} + c_1 + s_1 = x_{12} + f_1 \\ y_2 &= x_{21} + c_2 + s_2 = x_{21} + f_2 \end{aligned} \tag{19.37}$$

If we define coefficients $q_{12} = x_{12}/y_2$ and $q_{21} = x_{21}/y_1$, equations 19.37 can be written as

$$\begin{aligned} y_1 &= 0 + q_{12}y_2 + f_1 \\ y_2 &= q_{21}y_1 + 0 + f_2 \end{aligned}$$

which in matrix notation, using upper-case letters for matrices and lower-case for column vectors, is

$$\mathbf{y} = \mathbf{Qy} + \mathbf{f}$$

Table 19.10 Asset values for income groups and the world, $ per capita

Income group	Produced capital	Natural capital						
		Subsoil	Timber	NTFR	Cropland	Pastureland	Protected areas	Total
Low	1 174	325	109	48	1143	189	111	1925
Middle	5 347	1089	169	120	1583	407	129	3496
High OECD	76 193	3825	747	183	2008	1552	1215	9531
World	16 850	1302	252	104	1496	536	322	4011

Source: World Bank (2006)

with the solution

$$\mathbf{y} = (\mathbf{I} - \mathbf{Q})^{-1}\,\mathbf{f} = \mathbf{Lf} \tag{19.38}$$

where **I** is the identity matrix.

Now, let

$$D_1 = D_{M1} + D_{N1} = \mathrm{d}_{m1}y_1 + \mathrm{d}_{n1}y_1 = z_1 y_1$$

$$D_2 = D_{M2} + D_{N2} = \mathrm{d}_{m2}y_2 + \mathrm{d}_{n2}y_2 = z_2 y_2$$

so that we can write for total global depreciation

$$D = z_1 y_1 + z_2 y_2$$

or, in matrix notation

$$D = \mathbf{z'y} \tag{19.39}$$

where $\mathbf{z'}$ is $[z_1\ z_2]$. Substituting for **y** in equation 19.39 from Equation 19.38 gives

$$D = \mathbf{z'Lf}$$

or

$$\mathbf{T} = \mathbf{ZLF} \tag{19.40}$$

where **Z** and **F** are matrices with the elements of **z** and **f** along the diagonals, and zeroes elsewhere. For the two-country case, Equation 19.40 is:

$$\begin{bmatrix} t_{11} & t_{12} \\ t_{21} & t_{22} \end{bmatrix} = \begin{bmatrix} z_1 l_{11} f_1 & z_1 l_{12} f_2 \\ z_2 l_{21} f_1 & z_2 l_{22} f_2 \end{bmatrix}$$

In the matrix **T** the row elements give depreciation in a country arising by virtue of final demand in that and other countries, while column elements give depreciation in all countries by virtue of final demand in one country. So, row sums, D_i^{IN}, give depreciation in i, and column sums, D_i^{ATT}, give depreciation attributable to i. Thus, in the two-country case here $t_{11} + t_{12}$ is the depreciation of total capital actually taking place in country 1, while $t_{11} + t_{21}$ is the depreciation of capital in the global economy that is on account of, attributable to, final demand in country 1.

A slight extension of the method of Proops and Atkinson allows for consideration of these issues on a per capita basis. Let **P** be the matrix with the reciprocals of population sizes along the diagonal and zeroes elsewhere. Then, for the two-country case,

$$\mathbf{A} = \mathbf{TP} = \mathbf{ZLFP} \tag{19.41}$$

is

$$\begin{bmatrix} a_{11} & a_{12} \\ a_{21} & a_{22} \end{bmatrix} = \begin{bmatrix} z_1 l_{11}(f_1/p_1) & z_1 l_{12}(f_2/p_2) \\ z_2 l_{21}(f_1/p_1) & z_2 l_{22}(f_2/p_2) \end{bmatrix}$$

so that column sums from **A**, d_i^{ATT}, give depreciation in all countries attributable to per capita final demand in country i. And,

$$\mathbf{B} = \mathbf{PT} = \mathbf{PZLF} \tag{19.42}$$

is

$$\begin{bmatrix} b_{11} & b_{12} \\ b_{21} & b_{22} \end{bmatrix} = \begin{bmatrix} (z_1/p_1) l_{11} f_1 & (z_1/p_1) l_{12} f_2 \\ (z_2/p_2) l_{21} f_1 & (z_2/p_2) l_{22} f_2 \end{bmatrix}$$

so that row sums from ***B***, d_i^{IN}, give per capita depreciation in country i on account of global final demand. These depreciation measures can be compared with s_i, per capita saving in i.

The following interesting question can now be addressed: taking account of international trade, how does the average citizen of economy A compare with one of B in regard to contributions to the global difference between saving and the depreciation of total, man-made and natural, capital? This is an interesting question because, given trade, the sustainability question is really a global question – exhausting domestic natural resources is not a problem for a trading economy, provided that it has acquired other assets as it runs down its domestic resource stock, the income from which can replace its earnings from resource exportation. The problem really bites at the global level – the global economy is a closed economy which cannot import anything from anywhere. It is for this reason that most of the capital theory literature on sustainability, and the derived literature on accounting, deals with a closed economy.

To answer the question, we can calculate the elements of **A** and **B** above, and for each country use them to calculate the difference between its saving and its depreciation measured on the 'in' and 'attributable' basis. Some results are given in Table 19.11, where the entries are for the difference between $s_i - d_i$ for country i and $s - d$ for the global economy, where s is per capita saving and d is per capita depreciation. The upper part of the table refers to d_i calculated on the 'in' basis, the lower part to it calculated on the attributable basis. Clearly, for the global economy it makes no difference which way d is measured, and so there is just one row for $s - d$ in the middle of the table.

Table 19.11 Excesses of per capita saving over depreciation – difference from global excess

	$(s_i - d_i^{IN}) - (s - d)$ US$				
	1980	1982	1984	1986	1988
E. Europe	−79	−58	70	78	35
USSR	55	−76	289	278	285
W. Europe	570	341	344	522	764
Canada	838	808	760	525	953
USA	153	−200	38	−429	−401
Japan	1278	1377	1557	2603	4066
Oceania	349	11	62	−109	113
Africa	−102	−68	−113	−140	−238
Latin America	124	79	5	−66	−42
Other America	−142	−68	−363	−311	−206
Middle East	−578	853	−1024	−1135	−978
Other Asia	−132	−38	−67	−70	−163
s − d	**173**	**76**	**106**	**109**	**220**
	$(s_i - d_i^{ATT}) - (s - d)$ US$				
	1980	1982	1984	1986	1988
E. Europe	−53	−35	97	84	57
USSR	47	−100	276	266	286
W. Europe	440	249	306	528	754
Canada	984	1002	1020	774	1186
USA	48	−271	−141	−613	579
Japan	1123	1265	1512	2673	4110
Oceania	318	−45	62	−114	172
Africa	−102	−79	−119	−146	−246
Latin America	103	70	16	−66	−35
Other America	−158	−76	−236	−252	−236
Middle East	238	−273	−708	−950	−779
Other Asia	−139	−44	−70	−72	−161

The results in Table 19.11 use exactly the same data as Proops and Atkinson (1996) and therefore follow their categorisation of the global economy into 12 national and regional economies.[15] There are several interesting points about these results that are worth calling attention to. Note first that for the world as a whole, in each of the years distinguished, saving exceeded depreciation – genuine saving was positive. In noting this, we must also note that these data only cover the non-renewable resource component of natural capital. Overall, the picture in the upper part of the table is much the same as in the lower part – our appreciation of per capita national and regional contributions to the excess of global saving over depreciation is little affected by looking at things on an 'attributable' basis. Japan's per capita contribution is always greater than the global average, while that of Africa is always smaller. The situation for the USA is mixed, and in two years it does make a difference which way depreciation is measured. Note also that for the USA in every year except 1988, going from the 'in' to the 'attributable' basis for measuring depreciation reduces a positive entry or makes a negative one bigger.

Given our earlier discussion of the problems of measuring the depreciation of non-renewable resources, and the fact that this is the only part of D_N accounted for here, these results should not be invested with too much significance at the level of detail. The point is rather that the methodology developed by Proops and Atkinson provides an interesting perspective on the global sustainability problem. It can be seen as complementary to looking at the way consumption levels, and patterns, vary as between countries, and the arising implications for resources use, as reported, for example, in UNDP (1998). In that context, it should be noted that the results as presented in Table 19.10 take no account of ability to save – Japan and the USA, for example, have much higher per capita income levels than, for example, Africa and Other Asia.

19.5 Sustainable development indicators

In the first section of this chapter we looked at environmental indicators which are data relating to the natural environment. Sustainable development concerns what is happening in the economy as well as in the natural environment. In the previous three sections we have been looking at extensions of, and modifications to, standard accounting conventions for national income and wealth that seek to include changes in environmental conditions in such, monetary, measures. In this section we are going to look at examples of what we call sustainable development indicators, which are efforts, by official agencies and others, to provide data on the natural environment and the economy relevant to sustainable development other than via modified national income or wealth accounting. We look at just three

[15] We are grateful to John Proops for supplying us with the data.

such examples. References to other exercises of this nature will be found in the Further Reading section at the end of the chapter.

19.5.1 UK government sustainable development indicators

In 1994 the UK government adopted a strategy for sustainable development, and in 1996 it began to publish a set of indicators so that progress in that regard could be monitored. Both the strategy and the indicators have undergone a number of modifications over the subsequent years: information can be obtained from the website of the government department responsible for the strategy, the Department of Environment Food and Rural Affairs (DEFRA), http://www.defra.gov.uk. In 2005 the UK government published a new version of the strategy and identified four priority areas around which the publication of indicators would be organised. These were:

sustainable consumption and production;
climate change and energy;
protecting natural resources and enhancing the environment;
creating sustainable communities and a fairer world.

The most accessible source for UK government sustainable development indicators is the booklet *Sustainable development indicators in your pocket*, which first appeared in 2004. Tables 19.12 and 19.13 here are based on the 2008 edition of that booklet, DEFRA (2008b), hereafter SDIYP08. More information on these, and other, indicators can be found on the DEFRA website, from where the booklet can be downloaded.

The terminology in SDIYP08 is a little confusing in that an 'indicator' may comprise two or more 'measures', where a measure is data relating to a single phenomenon. There is no aggregation across the measures that go with an indicator. Thus, for example, the SDIYP08 entry for the first indicator listed in Table 19.12, Greenhouse gas emissions, gives information on two measures – there are time series for Greenhouse gas emissions (excluding aviation/shipping) and CO_2 emissions for 1990 to 2007. For the last of the 68 indicators, Wellbeing, 13 of indicators 1 through 67 are mentioned as being relevant to well-being, and data on an additional 12 measures are reported.

In the second column, Table 19.12 shows the Priority Area(s) to which each indicator is relevant. The third column gives a summary indication of the information provided in regard to whether things have been getting better or worse, or staying much the same, in regard to each indicator for which information is available, for the period 1990 to 2007. It must be stressed that these are only summary indications provided by the authors of this text on the basis of the information in SDIYP08. That information is the time series data on the measures (shown as graphs and diagrams in SDIYP08), and DEFRA's summary as √ for improvement, *x* for deterioration, and ≈ for no significant change in a measure. Of the measures reported on in SDIYP08, 17 are also reported as environmental indicators for the UK, as discussed in Section 19.1 above. In its reporting on environmental indicators, DEFRA does not use the √, *x* and ≈ summaries.

SDIYP08 is produced by the same government department as the UK's state of the environment report, which we looked at briefly in Section 19.1.2 above – see Table 19.2 for a list of environmental indicators for the UK. Given what sustainable development is about, it is not surprising that SDIYP08 uses many of the environmental indicators that appear in Table 19.2, in relation to the SCP, CCE and NRP priority areas, in addition to the CSC indicators. In terms of SCP, CCE and NRP – basically environmental and natural resource indicators – SDIYP08 contains most of the indicators from the state of the environment report, and introduces an indicator for resource use (the total mass of materials directly consumed), an indicator for minerals extraction for use in construction, an indicator for the difference between domestic energy production and consumption, and it gives more detail on energy consumption.

Table 19.13 summarises some of the information in SDIYP08 in terms of what it says about measures. There are 126 measures in all. Some measures relate to more than one priority area. The second column of Table 19.13 shows the number of measures relating to each Priority Area. In summarising what measures are saying, DEFRA ignores: measures that are

Table 19.12 Sustainable development indicators for the UK 2008

Indicator	Priority Area[a]	Progress[b]
1. Greenhouse gas emissions	SCP CCE	Mixed
2. Carbon dioxide emissions by end user	SCP CCE	Mixed
3. Aviation and shipping emissions	SCP CCE	Mixed
4. Renewable electricity	CCE	Positive
5. Electricity Generation	CCE	Positive wrt CO_2
6. Household energy use	SCP CCE	Positive wrt CO_2
7. Road transport	SCP CCE	Negative wrt CO_2 Positive wrt NO_x, PM_{10}
8. Private Cars	SCP CCE	Negative wrt CO_2
9. Road freight	SCP CCE	Negative wrt CO_2
10. Manufacturing sector	SCP CCE	Positive wrt CO_2, NO_x, SO_2, PM_{10}
11. Service sector	SCP CCE	Negative wrt CO_2 Positive wrt NO_x
12. Public sector	SCP CCE	Positive wrt CO_2, NO_x
13. Resource use	SCP	Positive
14. Energy supply	CCE	Negative
15. Water resource use	SCP	Positive
16. Domestic water consumption	SCP	No trend
17. Water stress	NRP	na
18. Waste	SCP	na
19. Household waste per person	SCP	Mixed
20. Bird populations	NRP	Mixed
21. Biodiversity conservation	NRP	
22. Agriculture sector	SCP CCE NRP	Mixed
23. Farming and environmental stewardship	NRP	Positive
24. Land use	NRP	na
25. Land recycling	SCP NRP CSC	Positive
26. Dwelling density	NRP CSC	Positive
27. Fish stocks	SCP NRP	Positive
28. Ecological impacts of air pollution	NRP	na
29. Emissions of air pollutants	SCP NRP	Positive for NH_3, NO_x, PM_{10}, SO_2
30. River quality	SCP NRP	Positive

Indicator	Priority Area[a]	Progress[b]
31. Flooding	NRP	na
32. Economic growth	SCP CCE NRP CSC	Positive
33. Productivity	SCP CSC	Positive
34. Investment	SCP CSC	Positive
35. Demography	SCP CSC	not stated
36. Households and dwellings	SCP NRP CSC	not stated
37. Active community participation	CSC	na
38. Crime	CSC	Positive
39. Fear of crime	CSC	Positive
40. Employment	CSC	No trend
41. Workless households	CSC	na
42. Economically inactive	CSC	No trend
43. Childhood poverty	CSC	Positive
44. Young adults	CSC	No trend
45. Pensioner poverty	CSC	Positive
46. Pension provision	CSC	No trend
47. Education	CSC	Positive
48. Sustainable development education	CSC	Indicator to be developed
49. Health inequality	CSC	Negative
50. Healthy life expectancy	CSC	No trend
51. Mortality rates	CSC	Positive
52. Smoking	CSC	na
53. Childhood obesity	CSC	Negative
54. Diet	CSC	na
55. Mobility	CSC	Mixed
56. Getting to school	CSC	No trend
57. Acessibility	CSC	na
58. Road accidents	CSC	Positive
59. Social justice	CSC	Indicator to be developed
60. Environmental quality	CSC NRP	na
61. Air quality and health	CSC	Mixed
62. Housing conditions	CSC	Positive
63. Households living in fuel poverty	CSC	na
64. Homelessness	CSC	Mixed
65. Local environmental quality	CSC	na
66. Satisfaction in local area	CSC	na
67. UK international assistance	CSC	Positive
68. Wellbeing	CSC	na

Source: DEFRA (2008b)

Notes: a SCP for Sustainable consumption and production
CCE for Climate change and energy
NRP for Natural resource protection and enhancing the environment
CSC for Creating sustainable communities and a fairer world

b Since 1990 or as far back toward then as data permits: DEFRA 2008b also reports progress since 1999

Table 19.13 Trends in UK sustainable development measures

	Measures	Used	Improvement %/Deterioration % Since 1990	Since 1999
SCP	49	30	70/20	66/10
CCE	23	14	50/43	43/36
NRP	29	24	58/8	67/4
CSC	67	51	39/11	47/8
All	126	100	49/16	53/30

Source: DEFRA (2008b)
Notes: SCP for Sustainable consumption and production
CCE for Climate change and energy
NRP for Natural resource protection and enhancing the environment
CSC for Creating sustainable communities and a fairer world

subsumed in other wider measures, measures driven by other measures already covered, and measures it classifies as 'contextual' (details at the DEFRA website). The third column of Table 19.13 shows how many measures are actually used as the basis for the fourth and fifth columns. The fourth column entry against SCP, for example, says that over the period since 1990 for 70% of the 30 measures used DEFRA gives them a √ and 20% get a *x*, so that 10% either show no trend or have no suitable information available. The fifth column gives the same information for the period since 1999.

Clearly, SDIYP08 contains a lot of information. Equally clearly, the information is hard to digest and summarise. There is in SDIYP08, or elsewhere in DEFRA's publications, no attempt to come up with a 'bottom-line' verdict on whether or not the UK's record is consistent with it realising sustainable development. Unlike the accounting approaches considered in the preceding sections, and unlike the approach to be considered next in this section, DEFRA does not aggregate across measures and/or indicators to come up with a single number indicator of performance. It explicitly rejects such an approach to monitoring progress on sustainable development:

> The indicators are first and foremost intended to communicate and highlight progress in key issues for sustainable development and for the priority areas, and along with other evidence to help identify where action is required.
>
> It may also be desirable to use the indicators to gain an impression of progress but it is not practicable or meaningful to combine all 126 disparate indicator measures into a single index of sustainable development. Aside from the technical difficulties involved, some indicator measures are more important or challenging than others and key messages would be lost.
>
> *(DEFRA 2008b, p. 9)*

The issues of weighting of components and aggregation across them in the construction of single number sustainable development indicators are examined in a survey of such indicators, Böhringer and Jochem (2007).

19.5.2 Measuring sustainable economic welfare

For most economists the weighting and aggregation of physical quantities using prices is so familiar that it is almost an instinct, and is not much reflected on.

In recent years a number of economists interested in environmental sustainability and economic welfare have taken the view that NDP is not the place from which to start the search for a satisfactory single number indicator. But, they have not abandoned weighting and aggregation using prices. Rather than NDP, they start with personal consumption expenditure as recorded in the national income accounts and then make a series of adjustments to it which are intended to produce a better account of welfare which is sustainable. The result is usually called an 'index of sustainable economic welfare' (ISEW) or a 'genuine progress indicator' (GPI). Figure 19.5 shows the results, in comparison with the movement of GDP, for the USA for the period 1950 to 2004.

In common with many ISEW/GPI results for industrialised economies, the main feature of Figure 19.5 is that whereas national income grew more or less continually over recent decades, 'properly' measured sustainable welfare grew much more slowly overall, and grew very slowly after the early 1970s; see, for examples, Daly and Cobb (1989) for the USA, Stockhammer *et al.* (1997) for Austria, and Hamilton (1999) for Australia. For the data of Figure 19.5, whereas GDP per capita grew by more than 200%, GPI per capita grew by just 75% over the whole period. The authors (Talberth *et al.*, 2007) who constructed the GPI data series shown in Figure 19.5 see their results as, like most ISEW/GPI estimates, supporting the threshold hypothesis generally attributed to Max-Neef:

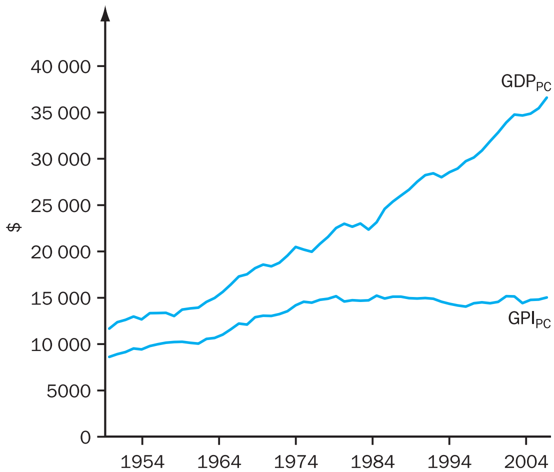


Figure 19.5 GPI per capita and GDP per capita for the USA, 1950–2004
Source: Data from Table 1 Talberth *et al*. (2007) units are constant (2000) $US.

> For every society there seems to be a period in which economic growth brings about an improvement in the quality of life, but only up to a point – the threshold point – beyond which if there is more economic growth, quality of life may begin to deteriorate.
>
> *(Max-Neef, 1995)*

The original ISEW was calculated for the USA by Daly and Cobb (1989), and we now briefly discuss their method and results. The definition that they use is

$$\text{ISEW} \equiv \{(C/D) + (E + F + G + H) - (I + J + K + L + M + N + O + P + Q + R + S + T + U) + (V + W)\}/\text{Pop} \tag{19.43}$$

where

C is personal consumption expenditure
D is an index of distributional inequality
E is an imputed value for extra-market labour services
F is an estimate of the flow of services from consumer durables
G is an estimate of the value of streets and highway services
H is an estimate of the value of publicly provided health and education services
I is expenditure on consumer durables
J is an estimate of private defensive spending on health and education
K is expenditure on advertising at the national level
L is an estimate of commuting cost
M is an estimate of the costs of urbanisation
N is an estimate of the costs of automobile accidents
O is an estimate of water pollution costs
P is an estimate of air pollution costs
Q is an estimate of noise pollution costs
R is an estimate of the costs of wetlands loss
S is an estimate of the costs of farmland loss
T is an estimate of the cost of non-renewable-resource depletion
U is an estimate of the cost of long-term environmental damage
V is an estimate of net additions to the stock of reproducible capital
W is the change in net overseas indebtedness.

Daly and Cobb (1989) report, in an appendix, the sources used and the estimation methods employed, and admit to the somewhat arbitrary assumptions that it was necessary to make in many cases. In effect, equation 19.43 is a welfare function, which reflects the authors' judgements about the determinants of welfare, and what sustains them. Others may have different views about these matters. It should be noted, for example, that according to equation 19.43, sustainable economic welfare increases when, other things constant, unpaid household labour increases. It should also be noted, on the other hand, that the value of leisure time does not appear as an argument in the welfare function. Many people would, one imagines, feel that their utility had improved if they did less work around the house and worked shorter hours in paid employment. That said, it should also be noted that per capita national income takes no account of time spent in paid work, nor of unpaid work.

Table 19.14 gives the results from calculations using the data provided in the appendix to Daly and Cobb (1989), which illustrate the sensitivity of the ISEW to the removal of some of its components. The two columns of numbers show the effect of changing, by one year, the base year from which the average annual growth rates for GDP, ISEW and ISEW variants are measured. Note that the difference between the growth of per capita GDP and

Table 19.14 GDP and ISEW average annual growth rates for the USA

	1950–1986	1951–1986
GDP	3.34	2.55
GDP per capita	2.02	1.52
ISEW	0.87	1.00
ISEW1	1.09	0.76
ISEW2 = ISEW1 − E	3.14	2.01
ISEW2 = ISEW1 + T	1.13	0.80
ISEW3 = ISEW1 + L + M + N	1.12	0.78

ISEW is reduced from 1.15% to 0.52%. ISEW1 is ISEW without the adjustment for the distribution of income; that is, it is the result if the term D in equation 19.43 is fixed at 1. The results for ISEW2, which is ISEW1 without the adjustment for unpaid extra market labour, shows that this adjustment has a major impact on the behaviour of ISEW. Without this adjustment, ISEW2 grows faster than GDP per capita whichever base year is used, and for the 1950 base ISEW2 grows by more than 1% faster. On the other hand, adding back in the adjustment for non-renewable-resource depletion, to get ISEW3, makes little difference to the growth rate obtained for ISEW1. As the last row shows, the non-renewable-resource depletion adjustment has an effect of essentially the same size as the adjustment for the costs of commuting, urbanisation and automobile accidents.

Other ISEW/GPI constructions make adjustments to personal consumption which, while generally similar in nature to those of Daly and Cobb, differ in detail on account of the judgements of the constructors and/or the availability of data. However, as noted above, they generally show similar patterns in relation to the behaviour of GDP per capita. Most ISEW/GPI constructions appear to have in common an adjustment for unpaid labour, and we have noted for the case of the Daly and Cobb (1989) ISEW the large leverage that this exerts on the final result. Neumayer (2000) argues for conventions for measuring environmental damage different from those used in most ISEW/GPI constructions and uses them in a sensitivity analysis, similar to that above, of ISEWs for four countries. He concludes that if there is a threshold, it is not due to movements in the environmental components of the index.

19.5.3 Efficiency ratios – measuring economic performance without using prices

One definition of economics is that it is the study of how human beings go about satisfying their needs and desires. As made clear in Chapter 2 of this book, doing that necessarily entails material and energy flows from the natural environment to the economy, and flows of wastes from the economy to the natural environment. The sustainability problem is now on the agenda because of a perception that these economy–environment interactions have reached levels which threaten the ability of the environment to continue to adequately support the economy.

Figure 19.6 is a simple representation of the 'big-picture' of what the economy does. It extracts materials and energy from its environment, using them along with labour and capital to produce the means to the satisfaction of needs and wants, and inserts back into the environment an equal mass of waste. Given this, and current concerns about sustainability, Common (2007a) proposes the ratio of satisfaction output to environmental input as a natural measure of economic performance. We can define

$$E = \frac{S}{I}$$

where E is for efficiency, S for satisfaction, and I for (environmental) input. Implementing such a proposal requires aggregation across many satisfiers to get a single number for S, and across many environmental extractions and insertions for I.

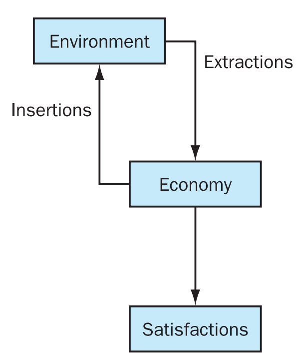


Figure 19.6 What the economy does

In the case of the numerator, S, we can look to the work on self-assessed happiness, or satisfaction, that was introduced in section 3.3.4 of Chapter 3 here. With H for the average score for self-assessed satisfaction and LY for average life expectancy at birth, a measure for S is:

$$\text{HLY} = H \times \text{LY}$$

Given that the economy–environment system is characterised by uncertainty, in the sense used in Chapter 13, we cannot predict the effects of current economy–environment interactions on the future path of HLY, and there is no uniquely correct way of aggregating across different environmental extractions and insertions. There are, however, partial proxy aggregate measures that can give useful information. Common (2007a) suggests three:

1. Energy use, because energy is necessary to do work and so its use is a measure of work done, which is the moving and transforming of matter, which is what impacts on the environment.
2. Ecological footprint, which, as discussed in Chapter 2, is the area of land and water required to provide an economy's environmental inputs and absorb its wastes, given current technology.
3. Greenhouse gas emissions, which are the source of the major environmental problem now facing the world, climate change, as discussed at various points in this book.

Using data from a variety of sources, Common (2007a) calculates five variants of the E ratio for 68 countries for 2000. There are five variants because energy use can be measured in two ways (excluding and including traditional non-marketed energy sources), as can greenhouse gas emissions (taking account of or ignoring the effect of land-use changes). Whichever way I is proxied, E shows a strong tendency to fall with per capita national income – richer countries tend to be less good at converting environmental impact into human satisfaction. Table 19.15 shows the five highest and lowest scores for E on the five different measures of I.

An increase in E represents improved economic performance but does not necessarily indicate sustainable development. With t for the start of a period and $t + 1$ for its end

$$E_t = \frac{S_t}{I_t} \quad \text{and} \quad E_{t+1} = \frac{S_{t+1}}{I_{t+1}}$$

Table 19.15 Highest and lowest E scores

E_{CE}		E_{TE}		E_F		E_{G1}		E_{G2}	
Country	HLY per ton[1]	Country	HLY per ton[1]	Country	HLY per ha	Country	HLY per ton Carbon[2]	Country	HLY per ton Carbon[2]
Bangladesh	336.00	Bangladesh	181.44	Bangladesh	56.00	Uruguay	501.00	Jordan	512.86
Senegal	104.33	Morocco	91.20	Vietnam	54.00	Bangladesh	168.00	Albania	113.00
Morocco	95.00	Philippines	64.77	Peru	45.89	Vietnam	144.00	Bangladesh	112.00
Honduras	94.2	Albania	62.85	India	42.63	Albania	84.75	Vietnam	108.00
Philippines	88.60	Peru	62.28	Morocco	42.22	El Salvador	71.86	El Salvador	100.60
Canada	7.41	USA	6.87	Latvia	7.55	Canada	9.23	Australia	8.81
USA	7.14	S Africa	6.77	Ukraine	7.48	Australia	8.69	USA	8.78
Luxembourg	7.00	Russia	6.68	Russia	6.43	Russia	7.65	Ukraine	8.52
Russia	6.74	Ivory Coast	6.04	USA	6.01	Estonia	7.35	Estonia	8.18
Iceland	5.39	Tanzania	3.50	Estonia	5.22	Zimbabwe	7.18	Russia	7.86

Source: Common (2007a)
Notes: E_{CE} I is commercial energy use
E_{TE} I is total energy use
E_F I is ecological footprint
E_{G1} I is greenhouse gas emissions including land use changes
E_{G2} I is greenhouse gas emissions excluding land use changes
1 toe for tonnes of oil equivalent
2 all greenhouse gases converted to heating equivalent CO_2

so that:

$$E_{t+1} \geq E_t \text{ for } \frac{S_{t+1}}{S_t} \geq \frac{I_{t+1}}{I_t}$$

An increase in E is consistent with both S and I increasing so long as S increases proportionately more than I. It is not obvious that it would always make sense to treat an increase in E that involved higher I as consistent with sustainable development. Saying that $E_{t+1} \geq E_t$ is consistent with sustainable development only if $I_{t+1} \leq I_t$ is not obviously sensible either. It would mean saying that poor countries with low environmental impact could not increase that impact and be judged to be developing sustainably, while a country with a very large but constant I and a small increase in S would be judged to be developing sustainably. For two variants of E there is an appealing way round this problem.

It has been suggested that the equitable way to set national greenhouse gas emissions allowances would be for each nation to be allocated an amount equal to its population size multiplied by an equal per capita share of whatever it has been decided is the current allowable total global emissions quantity. The ith nation's allocation would be $\text{GHG}_i^* = sP_i$ where P_i is its population and

$$s = \frac{\text{GHG}^*}{P}$$

is an equal per capita share of the global total GHG*, with P for the global population size. Then, we could say that country i experienced sustainable development over the period if $E_{i,t+1} \geq E_{i,t}$ and $\text{GHG}_{i,t} \leq \text{GHG}_i^*$ and $\text{GHG}_{i,t+1} \leq \text{GHG}_i^*$. An improvement in E would only be considered as sustainable development, that is, if at the beginning and the end of the period the country's greenhouse gas emissions were no larger than its equitable share of the global total for emissions. This would be the case for most developing countries, but would not be the case for most developed countries.

Looking now at using the ecological footprint for I, we could say that an increase in a country's E would only be considered sustainable development if $F_{it} \leq F_i^*$ and $F_{i,t+1} \leq F_i^*$ where F_{it} and $F_{i,t+1}$ are the country's footprints and F_i^* is its share of the world's available productive land, arrived at by multiplying an equal per capita share of that supply by the country's population size.[16] Again, for most developing countries $F_{it} \leq F_i^*$ and $F_{i,t+1} \leq F_i^*$ would hold, but for most developed countries it would not.

No country currently produces an official estimate of E. Most have produced official energy use data for many years, and are now required, by the UNFCCC, to produce regular estimates of greenhouse gas emissions.[17] Footprint estimates for most countries are now produced annually by the Global Footprint Network. Self-assessed satisfaction/happiness surveys are now conducted by various organisations in many countries, and a useful point of entry to the results arising is The World Data Base of Happiness, http://www1.eur.nl/fsw/happiness. Life expectancy data is readily available from a variety of sources. It should now be possible to begin to produce estimates of changes in E for many countries.

In the first sub-section here, 19.5.1, we looked at the UK government's activities in regard to sustainable development indicators. It is of some interest to note that SDIYP08 includes, at indicator 68 Wellbeing, data which are the results from self-assessed life satisfaction surveys for 2007 and 2008. These surveys were conducted on behalf of DEFRA and the Department of Health, and it is apparently intended to repeat them on an annual basis. SDIYP08 also contains data on life expectancy and greenhouse gas emissions. Unfortunately it is not possible to put all this together to come up with a number for the change in E for 2007 to 2008. This is, mainly, because the greenhouse gas emissions data end in 2007, and the life expectancy data in 2004/6. Going to other sources would not overcome this problem, but if the UK government does continue with its life

[16] Note that F_i^* and GHG_i^*, may change over time on account of changes in the size of the global population and/or changed appreciations as to the corresponding global totals.

[17] These greenhouse gas emissions data refer to emissions arising in a country. It is arguable that what is properly required is data on the emissions for which a country's consumption and investment are responsible, wherever they occur. Note that the ecological footprint accounts for land and water requirements wherever, within or outwith the country, they arise. On the other hand, greenhouse gas emissions estimates are likely to be more accurate than footprint estimates.

satisfaction surveys, it should soon be possible to derive an E based sustainable development indicator from official UK data sources. It is perhaps unlikely that DEFRA itself will do this, given its views on the limitations of a single index of sustainable development, quoted in 19.5.1 above.

19.6 Concluding remarks

Here we offer some brief remarks on where we think that we have got to, not just in this chapter but also in the course of the book as a whole. These remarks necessarily involve the authors' values and judgements, as well as technical economic considerations. The reader may come to different conclusions. Our purpose here is to offer our assessments for consideration, rather than to make pronouncements.

It will be clear that we do not see measuring sustainable national income, or genuine savings, as a practical step, or even an almost practical step, toward sustainability. Even if the capital-theoretic characterisation of the sustainability problem is accepted, and leaving aside the question of feasibility, it is our judgment that the practical problems involved are unlikely ever to be satisfactorily resolved. As well as the difficulty of ascertaining the 'right prices' for aggregation across quantities of reproducible and natural capital assets, there is the difficulty of measuring the quantities themselves across the full range of assets relevant to current and future human wellbeing. As noted in Chapter 2, in the case of biological species, many have not yet even been identified.

It is not even clear that measuring sustainable national income is a particularly desirable thing to do in pursuit of sustainability. The problem involved is multifaceted and complex and involves uncertainty. Dealing with it requires that these characteristics are recognised. Attempts to capture in a single number the answer to the question 'Are we behaving sustainably?' tend to obscure the essential characteristics of the problem. And, one of those essential characteristics is that it is, at bottom, a global rather than a national problem.

It is clear that economic analysis can contribute much to the discussion of other, in our view, more practical steps than trying to measure sustainable income, or genuine savings. We noted that the 'right prices' problem for that endeavour is something of a chicken and egg problem. If we are behaving as we should for sustainability, the right prices would be readily observable. If we are not, the prices that we do observe tell us little about how we should be behaving. However, while we may not know the 'right prices', so that computations of sustainable income are rather meaningless, we do in many cases have a good idea in which direction from current prices the right prices lie. In many contexts it is clear that using economic instruments – taxes, tradable permits and the like – would move actual prices paid by the users of environmental resources in the right direction. Making, for example, the use of fossil fuels more expensive would, we can be reasonably sure, do much for the amelioration of several of the environmental dimensions of the sustainability problem. In and of itself it could increase poverty, but that is a problem that can be addressed in other ways which economists are well equipped to advise on.

Of course, while economists can advise on dealing with the regressive impact of higher energy prices, this does not ensure that the appropriate measures will be put in place. That requires political action, as would the adoption of the fossil-fuel price-increasing measures in the first place. However good the analysis and the information, action will only occur if there is the political will for it. The hope is that good analysis and information will increase the political will to do 'the right thing'. It appears that for many, economist and non-economist, advocates of environmental accounting the real rationale for the activity is that it will produce results that affect the political climate. On this view, it does not really matter that the number produced is wrong or meaningless, so long as it moves perceptions in the 'right' direction. However, it is not clear that announcing in this way that economic performance is worse than had previously been thought, if that is the way the numbers do turn out, could be relied upon to have this effect.

On the other hand, we can be reasonably sure, on the basis of historical evidence, that actually changing prices does influence behaviour. It appears to us that, if the objective is to promote the cause of sustainability, it is much more important to move actual prices in the right direction than to get the right shadow prices for the computation of a number for sustainable income. To do this does require that decision makers and those who vote for them are well informed about the issues and the alternatives. Economists have an important role in providing some of this information, but there is an important role for other kinds of information, assessment and advice. Economists need to be honest about the quality of the information that they can provide, and the ethical basis for the advice that they offer. In saying this, we are not implying that economists are peculiar in this respect. All 'experts' contributing to public debate and deliberation on the many issues involved in the sustainability problem need to be circumspect about the limits of, and basis for, their expertise. It is simply that this book is addressed to students doing an economics course.

Summary

Environmental indicators

In many countries government departments and/or the national statistical office now regularly publish collections of biological and physical data concerning the natural environment. In some countries, such collections of environmental indicators are called State of the Environment Reports.

Environmental accounts

In many countries the national statistical office publishes monetary data concerning the natural environment. Such environmental accounts are organised around economic categories.

EDP

EDP is the term we use for national income measured so as to account for the environmental cost of economic activity. To date, most attempts to measure EDP have measured that cost just as the depletion of a limited range of natural resources.

Satellite accounts

The UNSTAT guidelines for environmental accounting do not recommend that national statistical agencies switch from reporting NDP to reporting EDP. They do recommend that environmental asset values, from which depletion could be inferred, are provided in satellite accounts.

Genuine saving

Genuine saving, equal to genuine investment, is the change in the value of all of an economy's assets. An economy for which genuine saving is positive at a point in time can be said to be operating sustainably at that point in time.

Sustainable development indicators

A variety of other sustainable development indicators have been proposed, not all of which can be properly be aggregated into a single comprehensive indicator which captures all dimensions of sustainable development.

Further reading

The capital-theoretic literature which is relevant to environmental accounting is extensive and growing rapidly. We shall here just note some of the major original contributions, and some recent papers that themselves provide fuller references to the literature. Dasgupta and Heal (1974) was perhaps the first rigorous consideration of the optimal path for consumption in a representative-agent single-commodity model, where the agent maximises the sum of discounted utility and production uses inputs of capital and a non-renewable resource. Solow (1974a) examined the feasibility of constant consumption for ever in such a model, given various assumptions about substitutability in production, population growth and technical progress. Weitzman (1976) established the interpretation of net national product as sustainable income as the return on wealth, but did not explicitly consider natural resources. Hartwick (1977) showed that for an economy with a constant returns to scale Cobb–Douglas production function with capital and a non-renewable resource as arguments, zero net investment – investment in reproducible capital equal to resource depreciation – would give constant consumption. Hartwick's rule was generalised, in terms of the production conditions in which it held, in a number of subsequent papers by himself and others: see for examples Hartwick (1978) and Dasgupta and Mitra (1983). Solow (1986) brought together the contributions of Weitzman and Hartwick, and set out the basic theory drawn upon in Solow (1992, 1993).

The literature on neoclassical growth theory, what we referred to as capital theory, in relation to sustainability was reviewed in Toman *et al.* (1995); see also Pezzey and Toman (2002), a Resources for the Future discussion paper available at www.rff.org/disc_papers/PDF_files/0203.pdf. Sustainability in small open economies is discussed in Asheim (1986) and Brekke (1997). Papers drawing on capital theory to derive propositions about the proper measurement of net national income as sustainable income, given the dependence of production and consumption on environmental inputs, include: Hartwick (1990), Mäler (1991), Dasgupta (1995), Hamilton (1994) and Cairns (2002). Volume 5, Parts 1 and 2, the February and May 2000 issue of *Environment and Development Economics*, is a special issue on 'Advances in green accounting' which includes papers on theory and practice. Faucheux *et al.* (1997) consider sustainability in the context of an overlapping generations model and argue that standard capital theory does not provide a satisfactory basis for environmental accounting. Weitzman (1997) looks at technical progress in relation to sustainability, and argues that on account of technical progress properly measured net national income will understate sustainable income.

Pezzey (2004) uses an optimisation model with exogenous technical progress and population growth to derive two measures that signal unsustainability, and provides a comprehensive bibliography. This paper explicitly mentions the 'paradox why sustainability should be of interest in a present-value-maximising economy'. If, that is, we accept that individuals should get what they want, and assume that they want pv maximisation, what is the justification for a sustainability objective expressed as a constraint? Pezzey's suggestion is that it is that individuals are dualistic (Pezzey uses 'schizophrenic'), choosing their own actions to maximise pv but voting for a government committed to taking care of the future. As Pezzey notes, such dualism has not been formally modelled as a basis for treating sustainability as a public good. Cairns (2008) distinguishes and discusses 3 concepts of income in the literature, deriving from different objective functions. Arrow *et al.* (2003) generalises the formal treatment of an imperfect economy as stated in Dasgupta (2001), and explores some partial models for the estimation of accounting prices.

The UNSTAT proposals for satellite accounting are set out in UN (1992, 1993b). Work on the UNSTAT proposals included the preparation of illustrative accounts for a hypothetical country, and of preliminary accounts for Mexico and Papua New Guinea, reported in chapters in Lutz (1993); see also Bartelmus (1994). Up-to-date information on progress with the proposals can be obtained by visiting the UNSTAT website at http://unstats.un.org. An 'operational manual', UN (2000), can be accessed electronically at http://unstats.un.org/unsd/sna1993/doc/F78e.pdf. The difficulties involved in actually measuring sustainable income are considered in Neumayer

(1999), see especially chapter 5. Hamilton (2000) reviews the theory of genuine savings measurement and reports 1997 results for some 150 countries. Young (1990) did some 'back of the envelope' calculations for Australia's EDP for 1980–88. Over the whole period he found that it (allowing for land degradation, timber depreciation, minerals depreciation and defensive expenditures) grew faster than GDP.

In the UK official data on the environment are now published bi-annually: at the time of writing the most recent environmental accounts available were in Office of National Statistics (2008a). Most of the data appearing in this publication now also appears annually in the Blue Book (Office of National Statistics 2008b). The Department for Environment Food and Rural Affairs produces an e-Digest of environmental statistics at http://www.defra.gov.uk/environment/statistics/index.htm. Two sources for environmental indicators for the EU are the European Environment Agency at http://themes.eea.europa.eu/indicators, and the European Environment Information and Observation Netwwork at http://eionet.europa.eu. In Australia state of the environment reporting is done by an independent panel and published by Department of the Environment, Water, Heritage, and the Arts: see Beeton *et al.* (2006). A very useful source of environmental indicator data, including energy and materials, for most countries of the world is the World Resources Institute, WRI: at http://www.wri.org click on Earth Trends. WRI also produces a Climate Analysis Indicators Tool, CAIT, which has lots of detailed climate change relevant indicators for most countries: go to http://cait.wri.org/.

Bleys (2008) calculates an ISEW for Belgium that involves some modifications to the formula set out in the chapter here, and finds that the time series does not conform to the usual ISEW/GPI pattern. Rather, it 'shows significant improvements in sustainable economic welfare over the entire study period'. Brennan (2008) argues that contrary to its critics, the ISEW does have a theoretical basis but that that basis is inadequate and needs to be developed by extension to incorporate a political economy analysis of the capitalist system in which the economies for which it has been calculated are embedded. Böhringer and Jochem (2007) look at 11 aggregate sustainable development indicators, including ISEW and environmentally adjusted economic accounts, in terms of the way they normalise, weight and aggregate their components. They conclude that 'such indices are doomed to be useless if not misleading with respect to concrete policy advice'.

In February 2008 President Sarkozy of France, expressing dissatisfaction with the state of information about the economy and society, set up 'The Commission on the Measurement of Economic Performance and Social Progress'. The trio leading this commission included two Nobel laureates in economics – Stiglitz and Sen. The Commission's report was published in September 2009 (Stiglitz *et al.*, 2009). The report noted some of the problems in measuring GDP, and argued that it had been asked to do what it cannot do – measure economic performance and social progress. It argued that no single measure can cover all aspects of economic performance, so that a suite of indicators are required. The commission recommended that national statistical agencies should report on other measures of well-being, including self-assessed satisfaction/happiness. It broadly endorsed the UNSTAT position on accounting for the environment, and recommended the compilation of biophysical data relating to the environment.

Discussion questions

1. Five European countries have access to the water resources of the River Rhine, which are intensively used for commercial and industrial purposes. Discuss (a) methods of valuation of Rhine water-quality degradation caused by human use, and (b) the allocation of these costs between the countries affected.
2. Discuss the arguments for and against the exclusion, or deduction, of defensive or preventive environmental expenditure from GDP. Identify other components of GDP which, it could be argued, should be excluded for identical or similar reasons.

3. Discuss the distinction between 'economic' and 'non-economic' environmental assets. Compile a short list of three or four specific non-economic environmental assets, and identify the costs and benefits associated with those assets, and how these might be valued for national accounts purposes.
4. There is a lot more coal remaining than there is natural gas, in the world as a whole. The combustion of coal releases more CO_2, and other pollutants, per unit energy released, than is the case with natural gas. Which should have the higher shadow price for the purposes of environmental accounting?
5. Devise a checklist for the qualitative and quantitative information which a university should be asked to furnish as a basis for an environmental audit of its functional activities.
6. Given the valuation problems inherent in assessing many forms of environmental damage or degradation, is it better to concentrate efforts on developing a comprehensive system of physical environmental accounts, rather than attempt to incorporate environmental costs and benefits into the conventional system of national accounts?

Problems

1. A mineral resource is extracted and sold, yielding £20m annual gross revenue to the owners. Purchases of goods and services used for extraction are £4m, labour costs are £2m and capital equipment is valued at £30m. The average rate of return on capital in the mineral extraction sector is 4.5%. At current extraction rates, reserves will be economically exhausted in 5 years. Assume a constant rate of extraction, a fixed extraction technology, and constant relative prices. Calculate a depletion rate for this mineral resource and hence the contribution of this extraction activity to gross and net national product, stating any necessary additional assumptions.
2. At the start of 1998 oil reserves in country *X* were 504×10^9 barrels. During 1998 country *X* produced 8×10^9 barrels, and there were no new discoveries of oil there. The world price of oil was constant at £3.125 per barrel throughout 1998, and the interest rate in *X* was also constant, at 5%. Total oil production costs in *X*, including a normal return on capital employed, were £20×10^9.
   (a) Calculate the depreciation of country *X*'s oil stock using
      1. the net present value method,
      2. El Serafy's user cost rule.
   (b) Repeat (a) using an interest rate of 10%.
   (c) Repeat the calculation for a 5% interest rate, but with the world price of oil being £3.00 at the start of the year and £5.00 at the end of the year.
   (d) Comment on your results.
3. In this chapter we showed that the owner of an oil deposit in a fully competitive economy would keep his or her wealth constant and achieve the highest consistent level of constant consumption by following the Hartwick rule. Show that this would also be the case for the sole owner of a fishery, given sustainable yield harvesting. Show that it would also be the case for someone owning an oil deposit and a fishery.

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Names Index

Subject Index