Handbook of Biodiversity Valuation

A GUIDE FOR POLICY MAKERS

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ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT

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GUIDE A L'INTENTION DES DÉCIDEURS

FOREWORD

In 1999, the OECD Working Group on Economic Aspects of Biodiversity (WGEAB) embarked on a project focusing on "…the monetary and non-monetary evaluation of the benefits of biodiversity and biological resources." This publication represents the main output of this project. It is supported by nine country case studies and by a compendium of related papers (OECD, 2001 – *Valuation of Biodiversity Benefits: Selected Studies*).

The Handbook is aimed at policy-makers and practitioners interested in using valuation tools for the effective management of biodiversity. Rather than offering an exhaustive catalogue of valuation methods, it emphasises only the major methodologies that are available, illustrated by examples. Given the OECD mandate, the primary emphasis here has been placed on the *economic* aspects of biodiversity valuation. This is in no way intended to deny the fundamentally cross-disciplinary nature of the issue. The Handbook recognises that concepts and methods drawn from various disciplines other than economics may also be appropriate for promoting the conservation and sustainable use of biodiversity.

In its recent *Environmental Outlook to 2020*, the importance of understanding the economic value of biodiversity for policy-making was emphasised. OECD Environment Ministers also made the economic valuation of biodiversity benefits a key focal point of their *Environmental Strategy for the First Decade of the 21st Century.* This Handbook responds to both of these interests. In a wider sense, it also contributes to the implementation of the Convention on Biological Diversity (CBD). Through its Decision IV/10, the Conference of the Parties (COP) to the CBD acknowledges that "economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures", and encourages the Parties to "take into account economic, social, cultural, and ethical valuation in the development of relevant incentive measures".

Under WGEAB's guidance, this Handbook was prepared by Professor David Pearce (Economics, University College London), Dr. Dominic Moran (Scottish Agricultural College) and Dr. Dan Biller (OECD Secretariat). Financial Assistance from the Government of France is gratefully acknowledged. This Handbook is published under the responsibility of the Secretary-General of the OECD.

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Biodiversity is valuable, as recognised by the Convention of Biological Diversity…

This handbook focuses on the nature of values associated to biological diversity (biodiversity) and the methodological approaches that can be adopted to assign values for policy purposes. It adopts a variety of case studies to illustrate the valuation process in OECD countries.

All societies depend on biodiversity and biological resources either directly or indirectly but their value is predominantly implicit rather than explicit. For biodiversity and many biological resources the absence of apparent value combined with absent or poorly defined property rights creates a problem of over exploitation and unregulated use. Increasing development pressures have led to an unprecedented rate of biodiversity loss. The resulting impacts on global well-being are sufficient to warrant a global convention - the Convention on Biological Diversity - to co-ordinate an international conservation effort.

... yet partly because much of the value is implicit rather than explicit, biodiversity continues to be lost at unprecedented rates.

While the Convention on Biological Diversity (CBD) stresses the role of concerted global action, the stark reality is that global action is only the sum total of actions taken within nation states that host our biological patrimony. Individual states and regions within states face conflicting priorities in the selection of development paths. Biodiversity conservation is often a low priority simply because there are measurement and valuation problems; biodiversity defies easy description and quantification. What cannot be quantified or is difficult to monitor and evaluate is easy to disregard. This adage also applies to the concept of value. While value has a variety of meanings it is manifestly true that the absence of an economic value for biodiversity and many

Biodiversity conservation is often a low priority because it is not easy to value.

biological resources means that they fail to compete on a level playing field with the forces that are driving their decline.

This handbook considers both economic and non-economic values of biodiversity…

This report emphasises the need to assign value to biodiversity as a prerequisite to an efficient approach to resource allocation. Biodiversity is a scarce and valuable global resource and conservation decisions must be taken to maximise this value within inescapable budget constraints. The volume is mainly though not exclusively concerned with the economic valuation of biodiversity. The importance of economic valuation is recognised in the CBD context. CBD's Conference of the Parties (COP) Decision IV/10 acknowledges that "economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures", and encourages the Parties to "take into account economic, social, cultural, and ethical valuation in the development of relevant incentive measures". While there are exceptions to the need to prioritise economic values over other cultural, traditional and spiritual values, the area of economic valuation has a sound theoretical foundation that can help clarify the tradeoffs implicit in public policy. Nevertheless, this volume does signal the limitations of an economic approach and considers how economic and non-economic values are related and can be reconciled.

… discussing what biodiversity is, the difficulties of measuring it, and the consequences of its loss.

In defining biodiversity Chapter II sets out the complexities inherent in the term and distinguishes between diversity and the biological resources that harbour diversity. The chapter highlights some of the difficulties in measuring the former but illustrates how some understanding of diversity can provide interesting insights for the design of an efficient conservation strategy. Data requirements for a consistent approach based on diversity measurement are formidable and biological resources (e.g. species and ecosystems) are adopted as the more manageable surrogate for conservation strategies. The chapter then considers the ecological consequences of biodiversity loss and evidence that suggests that loss is proceeding a historically unprecedented rate. A distinction between economic and non economic value criteria is introduced as the subject matter for Chapter III, which addresses some of the contrasting value systems being advanced in the global conservation debate.

Before detailing methodologies, the handbook discusses the different notions of biodiversity values.

The core of this debate concerns what may be conflicting stances on the relevant notion of value. For some people, the issue is about what is right or morally justified, and there may be only limited or negligible reference to cost and to what people may want. For others, what people want is itself a moral stance because of a presumption that providing what is wanted itself reflects a value judgement about the sensitivity of policy to wants - the 'democratic presumption'. Additionally, costs are very relevant because they represent the alternative use of funds and those alternative uses may themselves have moral content. There is no easy resolution of these different approaches and none is attempted in this manual. Those who favour the former approach will tend to want priorities for conservation sorted out by a legislature and a political process. Those who favour the latter will tend to opt for procedures such as cost-benefit analysis and multi-criteria analysis as prior requisites for what is ultimately always a political process.

Ultimately, whatever the value stance, a consensus exists around the imperative of safeguarding as much biodiversity as possible, subject to some consideration for the cost of doing so. Measured in terms of species, features or functions, this imperative embraces philosophical differences and establishes the minimum objective to one of cost-effectiveness of competing uses of a conservation budget. However budgets are determined, they should be used so as to maximise biodiversity conserved.

Cost-effectiveness analysis of conservation policy is however hampered by the fact that most programmes attempt to deliver multiple, frequently incommensurate outcomes. How these outcomes should be prioritised or weighted leads to another significant methodological divergence between approaches that use money or price weights and methods that use scores perhaps derived from expert group or public opinion. The latter weighting method characterises multi-criteria or multi-attribute modelling. The use of monetary weighting defines a cost-benefit approach to decision-making. The determination of monetary values for biodiversity is a central theme of later chapters of this volume. The derivation of these values allows biodiversity to compete on the same basis with other competing calls on public funding.

On valuation methodologies, the report discusses both non-monetary and qualitative decision-making processes.

Prior to expanding on this theme, Chapter IV addresses other qualitative decision-making processes that are also essential features of the philosophical debate. Complex environmental issues involve numerous stakeholders and many governments are responding to the call for more social involvement, public consultation and participation in policy decisions. Deliberative and inclusionary approaches seek to provide alternative arenas for eliciting social preferences. They do this by exposing a sample of the general public to the necessary scientific and social information to allow that group to reach a consensus position on a particular scientific priority or complex public policy issue. Citizen's Juries and Consensus Conferences are the most well known formats for this process and have become formal elements of decision-making in several OECD countries. For some the consensus process somehow provides a better or fairer reflection of social preferences rather than the more restricted private consumer model implicit in cost-benefit analysis. While participatory approaches can introduce other biases into decision-making, there is no reason to assume that they cannot themselves be used as an input to a more holistic cost-benefit test. Indeed, the two may be successfully combined.

Economic frameworks and specific valuation methods are discussed, including time discounting and how time preference rates may be adapted to take into account biodiversity issues.

Chapters V to IX concentrate in more detail on the economic framework and the specific valuation methods that allow biodiversity to enter into the cost-benefit decision-framework that is assumed to represent the conservation 'versus' development trade-off. Chapter V introduces the concept of time discounting and considers how time preference rates may be altered to account for the specific dilemmas faced by biodiversity conservation.

This is followed by an in-depth look at economic values and the economic methods available to assess them when markets fail.

Chapter VI spells out the economic interpretation of value and outlines the taxonomy of values associated with biodiversity. This range from direct use values associated with market prices through to non-use values that require more sophisticated enquiry methods to measure preferences not revealed in the market. The range of methodological approaches is then detailed in Chapters VI and VII, which discuss the range and limitations of economic valuation methods. The development of these methods is a fast moving research area for environmental economics, and their application to biodiversity presents particular problems related to the difficulties in identifying the nature of the good called biodiversity or in describing it to respondents.

A controversial but important tool - benefits transfer - is examined. It facilitates 'rapid appraisals' of biodiversity worth, but is not without methodological challenges.

Environmental valuation studies are generally time consuming and expensive to undertake and the number of possible values necessary for a complete understanding of the total economic valuation of biodiversity is likely to be large. In response to the urgent need for 'rapid appraisal' information some environmental economists have begun to consider the feasibility of borrowing results from existing studies and transferring them - suitably modified - to another similar site where information is needed. This practice is known as benefits transfer and is detailed in Chapter IX. Benefits transfer is not entirely new since cost-benefit appraisals have frequently transferred pre-existing externality values (e.g. a standard value of a statistical life is commonly used in different transport appraisals) for completeness. In the context of biodiversity, the process is arguably more complex. The process introduces a range of methodological challenges that make benefits transfer an interesting and evolving study area in its own right.

This Handbook should help policy-makers and practitioners to identify and implement successful biodiversity valuation methods, thereby furthering understanding of our common natural heritage.

Chapter X concludes the handbook by locating the cost and benefit information in a series of policy contexts ranging from land use planning to the determination of legal damages. The chapter reiterates the economic nature of the choices inherent in conservation policy and priority setting while considering some of the criticisms of a cost-benefit approach. An important caveat is that biodiversity conservation is characterised by a high degree of uncertainty. This means that whatever we learn from biodiversity valuation, a precautionary approach may still be needed to guide subsequent conservation or use decisions.

I. INTRODUCTION

1.1 Rationale

Though not always easily captured by markets, biological diversity (‘biodiversity’) is a valuable asset for current and future generations (OECD, 1999). Its conservation and sustainable use is one of the foundations of sustainable development and it is widely regarded as a priority environmental concern in OECD countries. In this context, the OECD underscores the importance of revealing the economic value of biodiversity in its recent environmental outlook (OECD, 2001 a) and identifies the valuation of biodiversity benefits as one of the pillars of the institution’s strategy and work (OECD, 2001 b). The importance of recognising biodiversity’s value and thus valuation tools is also enshrined in the Convention on Biological Diversity (CBD) agreed at the ‘Earth Summit’ in Rio de Janeiro in 1992. Through its Article 11, CBD calls on the Signatory Parties to “ … adopt economically and socially sound measures that act as incentives for the conservation and sustainable use of components of biological diversity”. Specifically regarding valuation, the Conference of the Parties (COP) Decision IV/10 acknowledges that “economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures”, and encourages the Parties to “take into account economic, social, cultural, and ethical valuation in the development of relevant incentive measures”. Valuation should thus be an integral part of biodiversity policies, one of the key requirements in devising conservation plans, and a basis for conservation and sustainable use. Moreover, assessments of value can help in raising public and political awareness of the importance of biodiversity.

In this context, this handbook continues the work of the OECD’s Working Group on Economic Aspects of Biodiversity (WGEAB) and its close collaboration with the CBD process. The importance of valuation is recognised throughout the Group’s decade long existence, but it is specifically tackled for the first time as an incentive in the Group’s first handbook (OECD, 1999). In its first handbook, out of twenty-two case studies provided to form its basis, four involved economic valuation as one of the incentive measures. These are available from the OECD's website (www.oecd.org). Following the guidance of the OECD environment ministers, WGEAB's agreed 1999-2001mandate, and the CBD COP Decision IV/10, WGEAB's second handbook focuses on valuation and is in part based on additional nine country case studies soon to be placed on OECD's website. Together with the previously published case studies, these case studies form a solid compendium of insights and practical experiences of policy makers from OECD member countries. The new country case studies include:

- Australia: Valuing Environmental Flows for Wetland Rehabilitation: An Application of Choice Modelling in the Macquarie Valley.

- Austria: Biodiversity, Landscapes and Ecosystem Services of Agriculture and Forestry in the Austrian Alpine Region - An Approach to Economic (E)Valuation.

- Canada: Application of Environmental Damage Assessment and Resource Valuation Processes in Atlantic Canada.

- Czech Republic: Applied Evaluation of Biodiversity.

- Hungary: Loss of Value of the Szigetköz Wetland due to the Gabčikovo-Nagymaros Barrage System Development: Application of Benefit Transfer in Hungary.

- Norway: The Norwegian Master Plan for Water Resources – A National Co-ordinated Plan for Non-Developed Hydropower Sources: Application of Multi-criteria Approach.

- Switzerland: Direct Payments for Biodiversity provided by Swiss Farmers: An Economic Interpretation of Direct Democratic Decision.

- United Kingdom: Valuing Management for Biodiversity in British Forests at the Forestry Commission.

- United Kingdom: Integrated Estates Management – Valuation of Conservation and Recreation Benefits.

1.2 Which value?

The importance of putting some value on biodiversity or at least establishing some kind of methodology to assign priorities in biodiversity–related issues and projects is widely recognised in different disciplines and sectors of society. All societies depend on biodiversity and biological resources either directly or indirectly but their value is predominantly implicit rather than explicit. For biodiversity and many biological resources the absence of apparent value combined with absent or poorly defined property rights creates a problem of over exploitation and unregulated use. Yet, individual states and regions within states face conflicting priorities in the selection of development paths. Biodiversity conservation and sustainable use is often a low priority simply because there are measurement and valuation problems; biodiversity defies easy description and quantification. What cannot be precisely quantified or is difficult to monitor and evaluate is easy to disregard. This adage also applies to the concept of value. While value has a variety of meanings it is manifestly true that the absence of an economic value for biodiversity and many biological resources means that they fail to compete on a level playing field with the forces that are driving their decline.

In defining biodiversity Chapter II sets out the complexities inherent in the term and distinguishes between diversity and the biological resources that harbour diversity. The chapter highlights some of the difficulties in measuring the former but illustrates how some understanding of diversity can provide interesting insights for the design of an efficient conservation strategy. Data requirements for a consistent approach based on diversity measurement are formidable and biological resources (e.g. species and ecosystems) are adopted as the more manageable surrogate for conservation strategies. The chapter then considers the ecological consequences of biodiversity loss and evidence that suggests that loss is progressing at a historically unprecedented rate. A distinction between economic and non economic value criteria is introduced as the subject matter for Chapter III, which addresses some of the opposing value systems being advanced in the global conservation debate.

The core of this debate concerns what may be conflicting stances on the relevant notion of value. For some people, the issue is about what is right or morally justified, and there may be only limited or negligible reference to cost and to what people may want. For others, what people want is itself a moral stance because of a presumption that providing what is wanted itself reflects a value judgement about the sensitivity of policy to wants - the 'democratic presumption'. In effect, the CBD process recognises the multi-character of value in biodiversity in COP's Decision IV/10 by encouraging the Parties to "take into account economic, social, cultural, and ethical valuation in the development of relevant incentive measures". Additionally, costs are very relevant because they represent the alternative use of funds and those alternative uses may themselves have moral content. While in many cases there is no conflict among the different views related to value, when conflict occurs there is no easy resolution of these different approaches. This is likely to remain a societal priority and the policy maker's choice reflecting the perceive priority. Those who favour the former approach will tend to want priorities for conservation sorted out by a legislature and a political process. Those who favour the latter will tend to opt for procedures such as cost-benefit analysis and multi-criteria analysis. Attempting to resolve such conflict is beyond the scope of this handbook.

1.3 Which valuation method to use?

Ultimately, whatever the value stance, a consensus exists around the imperative of safeguarding as much biodiversity as possible. Biodiversity is a scarce and valuable global resource and conservation decisions must be taken to maximise this value within inescapable budget constraints. Cost-effectiveness analysis of conservation policy is however hampered by the fact that most programmes attempt to deliver multiple, frequently incommensurate outcomes (OECD, 1999). How these outcomes should be prioritised or weighted leads to another significant methodological divergence between approaches that use money or price weights and methods that use scores perhaps derived from expert group or public opinion. The latter weighting method characterises multi-criteria or multi-attribute modelling. The use of monetary weighting defines a cost-benefit approach to decision-making. The derivation of these values allows biodiversity to compete on the same basis with other competing calls on public funding.

The need to assign value to biodiversity is thus a prerequisite to an efficient approach to resource allocation. Yet, the complex nature of biodiversity value results in an extensive menu of valuation methods available to policy makers and practitioners coming from different disciplines. Which method to use and under what circumstance are not simple questions to answer. Rather than providing an exhaustive catalogue of different valuation techniques, this handbook focuses on providing practical advice to policy makers and practitioners on the major methods primarily from economics. While there are exceptions to the need to prioritise economic values over other cultural, traditional and spiritual values, the area of economic valuation has a sound theoretical foundation that can help clarify the tradeoffs implicit in public policy. Moreover, this focus allows bringing together part of the emphasis of COP Decision IV/10, the OECD stated objectives and the comparative edge of the Working Group on Economic Aspects of Biodiversity. Through its well-targeted focus and its practical emphasis, it is hoped that the multi-character and cross-disciplinary nature of biodiversity benefits is harnessed rather than lost. In this respect, this volume does signal the limitations of an economic approach and considers how economic and non-economic values are related and can be reconciled.

Prior to expanding on economic valuation techniques, Chapter IV addresses other qualitative decision-making processes that are in some sense separate from the philosophical debate. Complex environmental issues involve numerous stakeholders and many governments are responding to the call for more social involvement, public consultation and participation in policy decisions. Deliberative and inclusive approaches seek to provide alternative arenas for eliciting social preferences. They do this by exposing a sample of the general public to the necessary scientific and social information to allow that group to reach a consensus position on a particular scientific priority or complex public policy issue. Citizen's Juries and Consensus Conferences are the most well known formats for this process and have become formal elements of decision-making in several OECD countries. For some the consensus process somehow provides a better or fairer reflection of social preferences rather than the more restricted private consumer model implicit in cost-benefit analysis. While participatory approaches can introduce other biases into decision-making, there is no reason to assume that they cannot themselves be used as an input to a more holistic cost-benefit test. In addition, they can be especially useful in awareness raising and education programmes. Through participatory procedures the 'value' of diversity can be revealed, even if not quantified.

The economic approach stresses the fact that any expenditure always has an opportunity cost, i.e. a benefit that is sacrificed because money is used in a particular way. For example, since biodiversity is threatened by many factors, but chiefly by changes in land use, measures of value denominated in monetary terms can be used to demonstrate the importance of biodiversity conservation relative to alternative uses of land. In this way, a better balance between 'developmental' needs and conservation can be illustrated. To date, that balance has tended to favour the conversion of land to industrial, residential and infrastructure use because biodiversity is not seen as having a significant market value. Economic approaches to valuation can help to identify that potential market value, whilst a further stage in the process of conservation is to 'create markets' where currently none exist. Market creation is the subject of a separate OECD initiative (OECD, forthcoming).

Chapters V - IX concentrate in more detail on the economic framework and the specific valuation methods that allow biodiversity to enter into the cost-benefit decision-framework that is assumed to represent the conservation versus development trade-off. Chapter V introduces the concept of time discounting and considers how time preference rates may be altered to account for the specific dilemmas faced by biodiversity conservation. Chapter VI spells out the economic interpretation of value and outlines the taxonomy of values associated with biodiversity. This ranges from direct use values associated with market prices through to non-use values that require more sophisticated methods of enquiry to measure preferences not revealed in the market. The range of methodological approaches is then detailed in Chapters VII and VIII, which discuss the range and limitations of economic valuation methods. The development of these methods is a fast moving research area for environmental economics and their application to biodiversity presents particular problems related to the difficulties in identifying market transactions for biodiversity or in describing it to respondents. Environmental valuation studies are generally time consuming and expensive to undertake since the number of relevant factors necessary for a complete understanding of the total economic valuation of biodiversity is likely to be large.

In response to the urgent need for information some environmental economists have begun to consider the feasibility of borrowing results from existing studies and transferring them - suitably modified - to another similar site where information is needed. This practice is known as benefits transfer and is detailed in Chapter IX. Benefits transfer is not entirely new since cost-benefit appraisals have frequently transferred pre-existing externality values (e.g. a standard statistical value of life is commonly used in different transport appraisals) for completeness. In the context of biodiversity, the process is arguably more complex. The process introduces a range of methodological challenges that make benefits transfer an interesting study area in its own right. Some OECD member countries are even establishing websites to facilitate the flow of information needed for benefits transfer (http://www.evri.ec.gc.ca/evri/).

1.4 What is this Handbook likely to tell you?

This volume is concerned with the ways in which value can be attached to biodiversity and, in particular, with the procedures and results of applying economic values. Non-economic procedures are also discussed in some detail. The advantages and disadvantages of different approaches are highlighted. The primary aim is to provide a convenient and pragmatic reference source to guide decision-makers and practitioners in the process of thinking about values in the context of biodiversity conservation and its sustainable use. The volume can be used to distinguish between types of value and their practical relevance to questions concerning biodiversity. It can also be used as a guide to the wider debate about how to value diversity and to some of the research that continues into the subject.

As a policy oriented report, this Handbook provides its policy recommendations in the last chapter. Chapter X concludes this Handbook by locating the cost and benefit information in a series of policy contexts ranging from land use planning to the determination of legal damages. The chapter reiterates the economic nature of the choices inherent in conservation policy and priority setting while considering some of the criticisms of a cost-benefit approach. It also places it in the context of the ecosystem approach, a strategy for integrated management of land, water and living resources supported by the CBD process. An important caveat is that biodiversity conservation is characterised by a high degree of uncertainty. This means that whatever we learn from biodiversity valuation, a precautionary approach may still be needed to guide subsequent conservation or use decisions.

As stated above, the focus on economic valuation is the result of a positive interaction among the recognition of its importance in policy making, complementary institutional goals and institutional comparative edge. Economic valuation methods should be used taking into account conditions specific to societies where they are applied and as indicated in this Handbook are by no means the only techniques available to researchers and policy makers. As with any other form of valuation, economic valuation has its limitations and drawbacks, which are analysed in both this volume and WGEAB's first Handbook (OECD, 1999). Regardless of its shortcomings, economic valuation plays an important role in educating decision-makers about biodiversity benefits, honing other incentive measures for biodiversity conservation and sustainable use, approximating the analysis of biodiversity issues to other policy issues that commonly use economic analysis as a basis, better informing policy makers regarding their choices under budget constraints, among other advantages. As with other incentives and other valuation methods, economic valuation should be an integral part of environmental policies and project analysis.

As with its first Handbook, the primary goal of this volume is to facilitate the task of policy makers in OECD Member countries in promoting biodiversity conservation and sustainable use and specifically in using valuation for this purpose. Yet, while based in part on OECD country case studies, the lessons drawn here can be adapted and applied in non-OECD countries as well. In fact, some techniques such as benefit transfer attempt to facilitate this process regardless of institutional affiliations. This flexibility makes the potential audience of this handbook larger, comprising all stakeholders involved in biodiversity management. This volume will be of interest and use to many groups, including NGOs concerned with conservation, public and private agencies investing in conservation measures, civil servants and academics, among others. The more stakeholders are informed about the tools available to them to sustainably manage biodiversity, the more likely biodiversity policies are to achieve their goals of conservation and sustainable use.

II. BIODIVERSITY LOSS AND BIODIVERSITY VALUE

2.1 Why ‘value’ biodiversity?

This Handbook is about the value of biological diversity and ways in which those values can be elicited. Exactly what is meant by ‘biodiversity’ is discussed in Section 2.2. A distinction is made there between *biological resources* and the *diversity* of those resources. Both the resources and their diversity have value. Those values may reside in the satisfaction that people get from using those resources, directly or indirectly, now or in the future, or in concerns that humankind has some wider responsibility towards other living things. The issue arises: why seek to categorise and measure, where possible, what those values are? The essential reason that values are important is that biodiversity competes with humankind for space on Earth. The value attached to conservation thus comes into conflict with the value attached to the uses of space that render biodiversity non-viable. When values conflict, it is important to understand just what the competing values are, to see whether some values are ‘more important’ than others, and to define and measure values that can be expressed in the same units as the activity that displaced biodiversity. While there can be no categorical presumption that the world has too much biodiversity, few would argue such a case, and the existence of a major international agreement on the need for its conservation - the Convention on Biological Diversity (CBD) - suggests that there is too little rather than too much. Further evidence that the rate of disappearance of habitats and species is faster now than for a very long period in the past, and that it is largely induced by human activity, lends weight to a widespread concern that whatever can be done to slow that rate of extinction should be done. Nonetheless, there are difficult trade-offs. Saving biodiversity is not a costless exercise. Choices have to be made and choices involve the balancing of values. This volume is about how economic valuation can help to inform rational policy decisions about biodiversity. The policy relevance of valuation information is extensive, but might include:

– demonstrating the value of biodiversity: awareness raising;

– land use decisions: for conservation or other uses;

– setting priorities for biodiversity conservation (within a limited budget);

– limiting biodiversity invasions;

– assessing biodiversity impacts of non-biodiversity investments;

– determining damages for loss of biodiversity: liability regimes;

– limiting or banning trade in endangered species;

– revising the national economic accounts;

– choosing economic instruments for saving biodiversity (e.g. taxes, subsidies).

2.2 Defining biological diversity

Any discussion of the value of biodiversity requires an understanding of what exactly the object of value is. A distinction needs to be made between *biological resources* and *biological diversity*. A biological resource is a given example of a gene, species or ecosystem. Biological diversity refers to the *variability* of biological resources, from genes to ecosystems. Thus, the CBD defines biodiversity as:

> 'the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part' (Convention on Biological Diversity, Article 2).

While the definition may be clear enough, measurement of biodiversity is very complex because diversity is multi-dimensional and something that defines complex systems. But work has begun to understand ways of making choices that involve trading off diversity. As an example, consider two habitats X and Y. In X there are six instances of one species S1, one of species S2, and one of species S3. In Y there are 4 instances of species S4, and four of species S5. A measure of *species richness* would suggest that X is more diverse than Y because it has more species. A measure of *species evenness*, however, would favour Y because there is less chance in Y that two randomly chosen instances will be of the same species (Purvis and Hector, 2000). Both richness and evenness are used as measures of diversity.

Biodiversity is the ‘variety of life’ whereas biological resources are the manifestation or embodiment of that variety. An example makes the distinction between resource and diversity of the resource clearer. In addition to richness and evenness, associated with the idea of diversity is the concept of ‘*distance*’, i.e. some measure of the dissimilarity of the resources in question. In the species context, efforts have been made to incorporate the concept of genetic differences between species in order to elicit the implications for conservation policy. One of the most important implications is that, without some idea of distance, it is very easy to conserve the ‘wrong’ set of species (or genes or ecosystems) if the aim is to conserve diversity. Solow *et al.*, (1993) provide an example in which information on the pair-wise distance between cranes and the extinction probabilities of those cranes. The situation in terms of extinction probabilities is shown below:

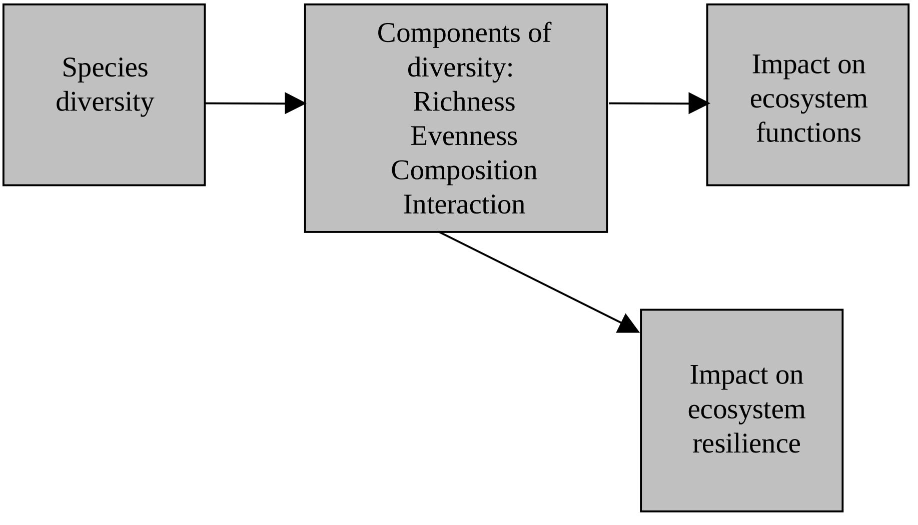
Endangered	Siberian	Extinction probability	0.9
	Whooping		0.9
Vulnerable	Japanese		0.7
	Hooded		0.7
	White-naped		0.7
Indeterminate	Black-necked		0.5
Safe	All others		0.0

The problem can be illustrated simply by assuming that budget constraints mean that only three species can be saved, and that the marginal costs of protection are the same regardless of the species in question**.** The issue is to determine which species should be saved. It is very tempting to allocate all resources to the most endangered species – the whooping and Siberian cranes. But if the policy objective is to minimise the (expected) loss of diversity, then Solow *et al.*, (1993) show that the optimal programme requires that the Siberian, white-naped and black-necked cranes should be conserved. Focusing on the most endangered does not in fact minimise biodiversity loss. The reason for this is that the genetic distance between the endangered species and at least one of the ‘safe’ species is small. Minimising the probability of the number of species lost is not the same as minimising the value of lost biodiversity. The example appears counterintuitive, not least because conservation policy often focuses on biological resources that are most threatened. Conservation policies that focus on very scarce biological resources may do so because they do not recognise the difference between biological diversity and biological resources. But policies may also be influenced by other factors, for example the values attached to scarce species may be high and conservation policy may be responding to those values. In short, conservation policy may be partly determined by the fact that people express values for endangered species and habitats rather than expressing values for the conservation of diversity *per se*. Nonetheless, if the stated aim is to conserve diversity, those policies may not be soundly based.

2.3 The ecological consequences of biodiversity loss

One approach to an appreciation of the value of biological diversity is to ask what the consequences are if significant losses of diversity occur. Clearly, the consequences depend in part on how diversity is defined - the notions of richness, evenness and distance have already been introduced as potential measures of diversity. Focusing on species diversity as the most studied expression of diversity, diversity can be said to affect *ecosystems* as shown in Figure 2.1.

Figure 2.1 Species Diversity and Ecosystems



According to Chapin *et al.,* (2000), most research has focused on the links between species richness (actual numbers of species) and ecosystem functioning. No clear links emerge from that work, perhaps because a few species only dominate in terms of ecosystem effects with the result that increasing the richness of species makes little change to the functioning of ecosystems. Species evenness may matter far more than richness. Species *composition*, i.e. the particular species that are present, is known to be important, as witnessed by the effects of introducing new species to ecosystems where they were previously absent. Similarly, the relationships between species

- *species interaction* - can affect ecosystem functions, for example by aiding nitrogen take-up by plants. So-called ‘trophic interactions’[1] are perhaps the best known linkages that, if modified, can result in significant ecosystem change. Removing predators can cause population explosions of the prey, which in turn results in loss of the food supply to the prey through over-exploitation.

The second main feature of ecological value in Figure 2.1 looks beyond the role of diversity in existing ecosystem function and points to the role that diversity plays in helping ecosystems ‘bounce back’ in the face of shocks or stresses. This is the *diversity-resilience* linkage. Ecosystems come under threat from various shocks and stresses, for example climate change. It is widely thought that systems that are more diverse have more capability to respond to such shocks, whereas those with low diversity are more likely to ‘collapse’ and not recover (Holling *et al*., 1994). In many respects this linkage is familiar from other contexts - someone saving for the future would adopt a portfolio of assets ranging from cash with no rate of return to long-term investments. The idea of having a portfolio is to spread risk so that events which threaten one asset are unlikely to threaten other assets. A diverse portfolio is therefore like a diverse array of species. Diversification of crops in farming adopts exactly the same idea and farmers may diversify even though it reduces overall productivity. The relevant ‘shocks’ in this context include local and global climatic change, but also cycles of pest invasions. There is evidence that, while the ‘green revolution’ has raised crop productivity substantially, to the benefit of human food supplies, it has also resulted in increased variability of output over time (Anderson and Hazell, 1989). More diverse systems may also be more resistance to species invasions (Chapin *et al.*, 2000). The diversity-resilience linkage gives rise to the notion of *an insurance value* of diversity. What is being insured against with more diverse systems is the risk that the whole system may collapse. More strictly, since risk tends to refer to contexts where probabilities of stress and shocks are known, the insurance is against uncertainty, i.e. a context where risks often are not known in any actuarial sense (Perrings, 1995).

Overall, then, changing species diversity, and especially changing evenness, composition and the linkages between species leads to changes in ecosystem function. Diversity also appears to have a strong role in conserving ecosystem functions in the context of external stresses and shocks from climate change to pests and exotic species invasions.

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[1] Trophic levels refer to the classification of organisms into producers (plants), primary consumers (herbivores), secondary consumers (carnivores and insect parasites) and tertiary consumers (higher carnivores).

To ecologists, part of the 'value' of diversity is that it preserves and regulates these functions: the ecological value of diversity shows up as its regulatory and protective role in ecosystem function.

Box 2.1 Use of environmental functions to communicate the values of a mangrove ecosystem under different management regimes

Complex ecosystems such as coastal mangroves are vital breeding grounds for birds and fisheries. They are under threat in many parts of the world for coastal development and pond fisheries. Such systems provide environmental goods that can be thought of as an endpoint to a complex production process that combines numerous ecological services to produce environmental functions. Sometimes more than one ecological service is thought to give rise to a function. Thus fuel (wood) production is a function of both the services of nutrient recycling and fixation of solar energy. A study by Gilbert and Janssen (1998) uses 110 hectares of mangrove forest at Pagbilao, Philippines as a case study. The study elaborates systems diagrams or models that envisage some of the many links that relate ecological processes to environmental functions, which in turn can provide services, some of which can be ultimately valued using economic techniques. The last stage provides the human interface, which is also the source of damage or stress to the ecosystem. In other words, use and development decisions within or beyond approximate sustainability constraints set off a chain process that leads to the perturbation of ecological processes. If it is possible to map the interconnectedness of ecological services and environmental functions (plus the feedbacks from use to original services) then a more comprehensive picture of the impacts of management options is feasible. The study attempts to identify and map these productive links as far as possible in system diagrams. The study also attempts to evaluate how the loss of vital links can impact the productive capacity of the system.

Having established the basic set of physical interrelationships the authors attempt to simulate the environmental effects of eight management options ranging from subsistence forestry through to commercial aquaculture. The resulting divergence of management impact from a base case (preservation) is measured both quantitatively in money terms for environmental goods and in most cases with a qualitative score (for most services).

The study shows how flexible modelling software capability can be exploited to explore complexity in the production of economic outputs by ecosystems. Simplifying assumptions are necessary in the face of ignorance. For example the linear relationships (linking stocks and flows) in the models are a necessary simplification of complex processes that may include discontinuities and hidden feedback effects. Such models can be refined and more fully calibrated as a better understanding of ecosystems emerges. The valuation of environmental endpoints is done using market prices, but the diagrammatic representation of how these values are produced and the basic services on which they most rely is informative for prioritising management options.

More anthropocentric notions of value are entirely consistent with the ecological notion of value: as indicated by Box 2.1 they simply take Figure 2.1 one stage further and ask what the effects of ecosystem change are on humans. The diversity-resilience link could be very important in this context since higher yields of crops could, in the extreme, be accompanied by higher risks of collapse of the underlying agro-ecosystem structure. Biotechnological developments may further impair ecosystem diversity in this respect.

2.4 Measuring diversity

The challenge in developing robust quantitative indicators of biodiversity lies in finding those that can be meaningfully applied for policy assessment. Here it is important to note that biodiversity is frequently discussed at different scales. An international debate about global extinction is often divorced from considerations of appropriate micro scales of analysis. Perlman and Adelson (1997) subject the shorthand CBD definition to a basic policy test of whether the categories can be used to assess the biodiversity of a region. Determining whether either a species or an ecosystem is present in a location is fraught with difficulties relating to fundamental definitional problems of what species and ecosystems are. More specifically, where does one species or ecosystem stop and another begin? The absence of any discrete cut-off point for determining boundaries between species (see Gaston and Spicer, 1998) or ecosystems is still subject to research and discussion. Even if this problem is overcome, the number of micro-organisms present at any location is likely to be staggering. Moving onto the genetic level, the numbers become even more unmanageable. The Human Genome Project gives some appreciation of the length of endeavour to map the genetic code of one species. Repeating the story for thousands more so that there can be some discrimination between them for policy purposes is a truly monumental task. Science has only a limited idea of the genetic dissimilarity between species. Taken together, these problems limit the use of the shorthand definition of biodiversity. As Perlman and Adelson put it:

> 'The current definitions of biodiversity as "genes, species and ecosystems" fail both in theory and in practice. First, they do not recognise the conceptual difficulties inherent in the constituent terms of biodiversity (namely genes, species and ecosystems). Second, they ignore the practical and technical problems involved in making real-world inventories of biodiversity. Third, they fail to take account of the incommensurabilities between different levels - how does one equate species with ecosystems, for example, in determining the biodiversity of an area? Finally, these definitions make no distinctions in the worth of

elements of biodiversity within any given level….' (Perlman and Adelson, 1997, 9-10).

Notwithstanding these definitional difficulties, the urgency to act focuses attention on the pragmatic use of biological information that is available in order to make a 'second best' approximation of the best conservation decisions. Although there is much interest in the development of indicators or inventories of ecosystem function, species richness is still the common approach to distilling the available information. Species richness is simply a systematic inventory of the number of species contained within an area. This is the commonest method for rapid impact statements about the change in diversity. In terms of approaches to valuation, species richness is also an easy concept to understand. Van Kooten (1998) notes that the measurement of biodiversity involves three aspects: scale, the component aspect and the viewpoint aspect. The scale element is made up of alpha diversity, beta diversity and gamma diversity. Alpha diversity is species richness within a local ecosystem. Beta diversity reflects the change in alpha diversity as one moves from one ecosystem to another across a landscape. Gamma diversity pertains to species richness at a regional or geographical level. This is a more global concept and a measure that is much more dependent on global shocks rather than the local ones (e.g. forest fires) that affect alpha and beta diversity. The component element of measurement concerns the identification of what constitutes a minimum viable population for the survival of a species. This is akin to setting *safe minimum standards* for species (see Chapter III). Finally, the viewpoint issue refers to the existence of many viewpoints, ranging from practical through to moral and aesthetic. Perlman and Adelson (1997) discuss the assignment of values in more detail. They note that viewpoints are necessarily subjective and value-laden (although, see Chapter III for one attempt to derive a 'value-free' approach), and that some value criteria have theoretical and legal standing irrespective of either their deliberate use or their ethical foundations (see Bockstael *et al*., 2000).

The derivation of quantitative indexes of biodiversity has preoccupied conservation biologists for several decades. The literature offers several ways forward in a range of diversity indicators using species richness ('alpha diversity') and evenness (the distribution of populations of various species within ecosystems). Magurran (1988) provides a wider treatment of some of these measures.

Two refinements can be made to the species richness indicator of diversity. One is to restrict the count to certain combinations of species only. The premise for this restriction derives from the literature on *surrogacy* (Williams and Gaston, 1994; Fjeldsa, 2000). This literature suggests that counting a limited set of certain species is a shortcut for representing overall diversity of other species that are not counted. As such, these surrogates are better indicators than those selected using other criteria – e.g. the physical appearance of species. Alternatively and perhaps additionally, species can be assigned importance weights using taxonomic information. Taxonomy is the field of science that looks at the relatedness or evolutionary difference between species using *family trees*. Such trees can be constructed using different units of account. The important point is that they show the difference between species based on the differences in the composition of these units of account. Thus, genetic divergence might be mapped to see how the genetic composition of a set of species diverged through time[2]. Species that have evolved differently will have a different genetic composition. The extent to which branches of a tree will have evolved with unique genetic histories gives an indication of the information value of the species currently identified as the end of the same branches. By the same token, and under some circumstances, longer and more remote branches provide information on the heaviest weights to be assigned to the species at the end of these branches.

In theory, taxonomic structures offer an interesting bridge between the economic and ecological literature on diversity measurement. If nothing else, taxonomic data are information and information has value. Contributions by Weitzman (1992) and Solow *et al*., (1993) show how this information can be useful for prioritising conservation spending. Crucially these insights assume the existence of complete taxonomic histories (trees) for the species of interest, as well as an understanding of the extent to which a successful intervention can reduce extinction probabilities for the branches over a given time. Combining the probabilities with the information on intrinsic uniqueness embodied in taxonomic structure, Weitzman derives a method for prioritising interventions among species represented by a cladogram. The method reduces to a cost-effectiveness criterion (cost per avoided expected loss of diversity), by considering what part of the tree should be the focus of conservation expenditure. The implications of this analysis clarify thinking on how to prioritise between a species that is say, moderately endangered but rich in unique history, and one that is highly endangered but genetically[3] relatively

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2 The technical terms for this is a cladogram. This is a tree-like graphical representation of how species come into being through evolutionary time and the relationship between terminal taxa, which may be species but can also be higher taxa such as orders or families.

3 Genetic information is used by way of example, but the result can be generalised provided taxonomic information is available to map out the accumulation of any distinguishing trait.

poor. The approach illuminates the inefficiency of sweeping conservation programmes that fail to recognise the inevitability of some extinction and that allocate all resources to the most endangered species at the expense of everything else.

Other ecological principles emerging from the taxonomic set selection literature have analogous economic interpretations that can be applied at the less exacting level of species richness conservation strategies. Pressy *et al*., (1993) emphasise the corollary issues of *complementarity, flexibility* and *irreplaceability*. Complementarity stresses efficiency in coverage, or the idea that conservation should avoid duplication or redundancy in conserved sites by selecting sites that harbour things that have not already been protected elsewhere. In the context of taxonomic information, this implies conserving the collection of species that corresponds to maximising the length of tree branches that are unique. Flexibility means that among a group of non-unique solutions for conservation strategies, one should choose the least cost solution. Irreplaceability stresses the number of sites that are the unique options for acting as reserves. These then operate as constraints on any other choice criteria. This criterion re-emphasises the need for a precautionary approach to selecting reserve areas in the face of irreversibility.

Taxonomic set selection is extremely data demanding. Comprehensive taxonomic information is unavailable for most species and the precision of such a weighting exercise is limited by the cost of collecting data. Monitoring cost and decision urgency are issues when considering any level of diversity. Higher order classifications (e.g. species richness and landscapes) are less costly to identify and monitor but offer less precision in terms of their approximation of lower level diversity[4]. At a general level, the relationship between cost of monitoring and the precision of the biological unit of account is given in Table 2.1.

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[4] The caveat to this being that a literature on surrogacy is currently discussing the extent to which species richness can act as an adequate surrogate measure provided the right combination of species is targeted.

Table 2.1 The biological hierarchy and monitoring costs

Precision and cost (of measurement) as a measure of character diversity	A scale of surrogacy for character diversity
Low (precision and cost)	(Ecosystems)
↓	Landscapes
↓	Land classes
↓	Species assemblages
↓	Higher taxa
↓	Species
High (precision and monitoring cost)	(characters e.g. genetic complement))

Source: adapted from Williams and Humphries (1996).

Cost constraints aside, conservation objectives can be highly subjective. When decision-makers refer to biodiversity they are referring to a classification structure that is shaped by their own values and interests rather than to any value-free entity. Some decision criteria are more transparent than others.

Decisions based on economic value are informed by a particular anthropocentric value criterion based on observed and verifiable trade-offs made by humans. Such trade-offs can be motivated by different reasons that go beyond mere self-interest. Most trade-offs are informed by known consequences. Biodiversity is one case where the consequences of choices are unclear. Bad choices can be made in the absence of information about how they may ultimately affect human well-being. Conversely, good decisions can be made if they are based on an anticipated improvement in human well-being. Economic valuation attempts to observe or emulate this trade-off scenario and to provide good information to those making the choices.

Although it is important to recognise the limitations of methods set out in this Handbook, economic valuation has merit in terms of the explicit use of a monetary numeraire to measure the efficiency of conservation. This helps determine where the greatest return to conservation spending can be obtained. The concept of efficiency as applied to conservation decisions is a unifying theme for economists and conservation biologists. The difference is that the latter group does not use explicit monetary valuation. Later chapters focus on cost-effectiveness and other approaches.

2.5 Valuation and the Convention on Biological Diversity

- The Convention on Biological Diversity (CBD) seeks the following goals:

  - The conservation of biological diversity.

  - The sustainable use of its components.

  - The fair and equitable sharing of the benefits arising from the use of genetic resources.

Explicit and implicit in these goals is the notion that biodiversity has value globally and locally. As indicated in the introduction, the CBD process takes on the issue of value and valuation further through the Conference of the Parties (COP) Decision IV/10, which acknowledges that "economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures" and encourages the Parties to "take into account economic, social, cultural, and ethical valuation in the development of relevant incentive measures". The very first issue that arises, then, is the quantification of value. If the world is to devote more resources to biodiversity conservation an obvious question is how much extra should it devote? There can be no answer to this question without some idea of whether the value received in return for a unit of expenditure is 'worth' that expenditure.

A driving force behind the CBD, however, is the fact that a very large part of the world's biodiversity resides in the poorer countries of the world, i.e. in those countries least able to finance its conservation and least able to resist the land use changes that threaten biodiversity. The CBD thus contains two compensating mechanisms. The first involves the richer world allocating 'new' resources to the financing of conservation in the developing world, in addition to those efforts that they make in their own countries. The second involves ensuring that developing countries gain a more equitable share in the financial and other benefits that the rich world derives from the biodiversity of the poor world. These factors point to the second area where the issue of value has to be debated. What flows of resources from rich to poor countries would be justified in the interests of helping developing countries conserve their biodiversity? Unless there is some idea of the value that the world as a whole gets back, and, indeed, what the donor countries get back, from such investments, the question of what resources to transfer is likely to be settled on an *ad hoc* and probably unsatisfactory basis. In the CBD the issue is linked to the notion of 'incremental cost', the extra cost of changing a management practice or a policy or an investment so that it generates global benefits, i.e.

benefits to the rest of world outside the country in possession of the biodiversity. Essentially, the value of the global benefit that is secured must exceed the incremental cost of making the change.

The third area where value is relevant, concerns the notion that *sustainable use* of biodiversity is a goal for all countries. It is well known that, if the time horizon for using a resource is fairly short-term, *unsustainable* use will often be of greater benefit than sustainable use. This is true, for example, of sustainable forest management compared to conventional logging, and it is true of forest conversion to agriculture compared to forest conservation (Pearce *et al.*, 2001). Sustainable use has more justification when the time horizon is very much longer[5] (see Chapter V). But the poorer someone is, the less likely it is that they will look far into the future, although poverty alleviation and conservation can often go hand-in-hand. To the impoverished slash and burn agriculturist, sustainable use is hardly an option[6]. Hence, persuading those on the margins of poverty to switch into sustainable use systems involves compensating them for foregoing of short-run gain, even if they themselves gain in the longer run from sustainable management systems. The relative values of short and long run concerns have to be changed.

Finally, the CBD insists that there is a fairer sharing of the value derived from biodiversity. What constitutes a fair share is the subject of a substantial philosophical and practical literature. But the notion of a 'fair return' to the 'owners' of biodiversity is clearly relevant, and fair returns are about values.

Unsurprisingly, the issue of conflicting values and the size of the value of biodiversity pervade the CBD and the principles enunciated in it, even if it is not expressed in these terms. Swanson (1997) states:

> '(The Biodiversity Convention) … has been concluded by virtue of the coalescence of a wide range if disparate interests, all concerned with the same general underlying problem: the increasing homogenisation of the world and the failure to invest in the diverse…. the underlying concern is the same: the absence of any systematic approach to encouraging investment in the value of diversity' (Swanson, 1997, p.18).

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5 And when the discount rate is low, see Chapter 4.

6 In times past it was, burning being followed by lengthy fallows and regeneration of the land.

2.6 Rates of biodiversity loss

The number of species in the world is not known. The number of *described* organisms totals some 1.75 million, and it is conjectured that this may be just 13% of the true total, i.e. actual species number perhaps 13.6 million (Hawksworth and Kalin-Arroyo, 1995). The discovery of new species is in fact not uncommon (Purvis and Hector, 2000), but the general focus is rightly on the *loss* of biodiversity. There are several ways of looking at biodiversity loss. The average 'age' - i.e. the time a species has been on Earth - of extinct species is around 5 million years. If there are 13.6 million species, then 13.6/5 = 2.75 species can disappear each year without total diversity shrinking. Yet, it is known that the loss rate is substantially greater than this since *documented* species losses since 1600 have been around 2.8 per year, and the rate is increasing (Purvis and Hector, 2000).

While still controversial, species-area relationships, which predict the number of species lost based on the area lost, suggest that loss rates run into the thousands per year[7]. Numerous studies have been made on tropical forest extinction rates. Using a species-area relationship, assuming that tropical forests account for about one-half of all species diversity, loss rates of tropical forest of just under 1 per cent area per annum would result in 1-10% of the world's species being lost over the next 25 years (Barbault and Sastapradja, 1995). The species-area relationship also entails that current rates of conversion of 'natural' areas will not result in very rapid rates of species loss compared to the loss rates that will ensue when yet further land conversion occurs. In other words, loss rates build up rapidly as the area in question is reduced: 'fewer extinctions now, many more later' (Pimm and Raven, 2000).

This situation is exacerbated by the concentration of much diversity into 'hotspots' where rates of land conversion tend to be highest. Moreover, there is a delayed impact of area reduction on species loss. Even if all remaining hotspot land was immediately protected, 18% of their species will nonetheless disappear. If only currently protected hotspot areas remain in a decade's time,

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7 The species area relationship takes the form $S = cA^x$ where S is the number of species, c is a constant reflecting the density of species per unit area, A is area and x is the slope of the relationship between S and A when S and A are expressed as logarithms. Low values of x indicate that considerable amounts of area can be lost without dramatic effects on species loss - e.g. for x = 0.15, 60% of area lost would result in just 13% loss of species. But the curve then rises dramatically so that the next lost of area results in disproportionately more species being lost. Otherwise, the higher the value of x, the greater the species loss for any given loss of area.

40% of hotspot species will disappear (Pimm and Raven, 2000). This explains why the hotspots approach to conservation has proved compelling: the species-area linkage applied to areas of high species richness and major rates of land conversion indicates a very rapid rate of loss of species unless there is a dramatic programme of protection. As noted below, however, the cost of such programmes may be high and their chances of success low, making the biodiversity problem particularly perplexing.

2.7 Setting priorities for conservation

Determining priorities for conservation and/or sustainable use of biological resources and diversity is essential. These resources are under threat all over the world. Hence the policy initiatives that would need to be mounted even to conserve what remains would be formidable. Resources for conservation are limited, for whatever reason, so that setting priorities is important. As noted above, the priority setting will probably differ if the aim is to conserve diversity rather than resources. But even if the aim is to conserve biodiversity, priority setting is complicated because it does not necessarily follow that resources should be allocated first to the scarcest or most threatened biodiversity, even though this is a widely recommended procedure (Myers *et al*., 2000). Prioritising action according to the degree of threat of extinction could ignore the reason why the biodiversity is severely threatened in the first place. If the cause of extinction is not very amenable to policy measures, allocating resources to conservation is likely to be wasteful anyway. This suggests an approach based on cost-effectiveness rather than scarcity, on securing the largest amount of conservation for a given level of expenditure (Moran *et al*., 1997; Cracraft, 1999). The kinds of issues that would need to be taken into account into any priority indicator would include the degree of scarcity and the concentration of diversity, but also the chances that an intervention will succeed. Those chances will depend on what factors are responsible for the degree of threat, and on the demonstration of commitment to conservation by the relevant local agencies and by government. Some causal factors may not be amenable to policy interventions: very rapid population growth for example. Others, such as misdirected policies that result in the socially uneconomic conversion of land areas, may be far more easily corrected.

As suggested by Box 2.2, priority setting is therefore more complex than identifying ‘hotspots’ - geographical concentrations of large numbers of endemic species under serious threat - and involves careful assessment of the costs of intervention, the nature of the threats and their amenability to policy measures, and the risks that interventions might not be sustained by the relevant local agencies. The assessment of values can assist this process but is only part of the overall package of measures that will be required.

Box 2.2 Determining priorities for biodiversity conservation expenditures

It is widely agreed that biodiversity conservation should be cost-effective. Cost-effectiveness involves estimating the cost of any action or set of actions to save biodiversity, and relating this cost (K) to some measure of biodiversity conserved (B). The latter involves the use of some indicator of biodiversity, such as species richness. But a simple ratio B/K is misleading. Biodiversity is under various degrees of threat and policy interventions may well be unsuccessful if the threat is one that is not amenable to policy. Similarly, the success of an intervention will depend on the commitment of the local and central governments involved. Several attempts have been made to combine all these factors – cost, the measure of biodiversity saved, the degree of threat and the chances of success - into a single index. One such attempt by Moran *et al*., (1997) produces an index that takes the form:

$$E = [A.(1-k)^n.B]/K$$

where E is cost-effectiveness, A is the percentage of protectable area that is protected, k is the rate of growth of the threat (e.g. deforestation, population change), B is the change in biodiversity due to the intervention and K is cost. The value of n is given by the period over which the past success of interventions is measured. Applied to the Asia-Pacific region, the index suggests the following ranking of priorities for conservation effort (selection only):

Country	Rank	Index value
Pakistan	1	12.8
China	2	0.8
Bangladesh	3	0.8
Sri Lanka	4	0.5
Vietnam	5	0.5
Thailand	6	0.5
India	7	0.3

The significance of the approach is seen by comparing the index with those based on a straight species richness criterion, the so called megadiversity or ‘hotspot’ regions. In megadiverse terms, Indonesia would be ranked first, followed by Malaysia, but neither country is in the top seven of the Moran *et al*. Index. In terms of hot spots, parts of Indonesia would again be ranked first, followed by Malaysia.

Earlier chapters have indicated that the proper context of biodiversity conservation is one of priority setting. Not all biodiversity can be conserved. Hence there have to be choices about what interventions are to be made and what, by implication, must be sacrificed. Priority setting is complex since it involves a number of issues:

a. which measure of diversity is to be used;

b. the degree of threat to that diversity;

c. the immediacy of any threat;

d. the chances that any intervention will be successful.

In essence, the search for an indicator to accommodate this information amounts to an attempt to introduce a cost-effectiveness approach to conservation. Cracraft (1999) suggests several measures, which encompass these features. This is but one of a number of approaches and it can be used to illustrate the general procedures involved.

Measures of diversity

The Species Diversity Index (SDI) consists of rankings by country according to diversity in higher plants, butterflies, land birds and mammals. The rankings are then summed to produce a score for each country, on the assumption of equal weights for the four indicators, and ignoring correlations between butterflies, mammals and bird species.

Measures of threat

Cracraft's (1999) Biodiversity Threat Index (BTI) adopts four measures and secures a ranking of each threat category for each country. As with the SDI, rankings are summed on the assumption that each category is equally important. The categories are population density, percentage of land area subject to high disturbance, change in cropland area and annual percentage forest loss. The focus on land use change is important since it reflects the consensus view that biodiversity loss is mainly due to that factor.

Measures of potential success

To deal with the potential for success in conservation, Cracraft proposes a Capacity Response Index (CRI). The assumption is that higher economic and social development corresponds with a higher capacity to respond to biodiversity threats. The index used is the Human Development Index produced by the UNDP and which in turn is an amalgam of education/literacy variables, life expectancy and adjusted GNP.

The SDI, BTI and CRI can be compared. For example, Cracraft (1999) finds that, in tropical America, three countries occupy the highest combined threat and high diversity categories: Ecuador, Venezuela and Mexico. Comparison of the CRI and BTI indicates that Venezuela and Mexico also have the highest capacity to respond (in tropical American countries) and Ecuador has a medium CRI. Similar classifications can be built up using global data. The policy implications might be formulated as follows:

a. countries with low threat and low diversity have traditionally not been seen as priority concerns, those with high threat and high diversity have been;

b. once capacity to conserve is considered, countries with low capacities and low threat are not priorities, those with high threat and high capacity are priorities.

As Cracraft notes, however, it is possible to advance the case that priority areas should be those with high threat and low capacity. It also matters who is responsible for financing the policy initiatives: if the funding is internal to the country in question then high capacity is a good sign. If the issue is one of foreign aid, low capacity may be more relevant. Finally, cost considerations are only implicit in the analysis (via development status) whereas costs of conservation vary widely between nations. Moran *et al.*, (1997) specifically include conservation costs in their approach to the issue. They combine measures of diversity with threats, country performance in conservation, and costs. Comparisons of the Cracraft and Moran *et al*. indicators suggests that fairly similar priority rankings are secured.

2.8 The economic consequences of biodiversity loss

The economic consequences of biodiversity loss follow from Figure 2.1. It shows that there are two broad ecological consequences. First, some ecosystem functions may be lost and, second, the resilience of the whole system may be impaired. Clearly, these two effects are interrelated.

Loss of ecosystem function

By and large, all ecological functions of ecosystems are economic functions since humans make use directly or indirectly of all ecosystems. The challenge for economics is that a great many of these uses, e.g. climate regulation, do not have markets. Hence, what appears on the financial balance sheet are low money values for diversity because so many of the effects of changing diversity have no markets in which financial values are revealed. This is the familiar problem of non-market distortions and later chapters discuss the procedures for eliciting these non-market values.

The direct relevance of ensuring that economic values for non-market ecosystem effects are recorded lies in the judgement made earlier that most diversity loss is due to land use change. In turn, land use change is primarily driven by the respective rates of return to the different land uses. A forest converted to agriculture appears to have a higher economic value than as a conserved forest. 'Green belt' land in richer countries appears to have low conservation value relative to the value of the land for housing developments, and so on. While economic values may not capture by any means all of the 'value' residing in diversity, the importance of economic value derives from its role in altering the accounting balance sheet for land conversion. The higher non-market economic values are, the less likely it is that land conversion that damages biodiversity will be justified. The corollary is that simply measuring non-market values is not enough: they have to be 'captured' through some process that converts non-market values into real financial or resource flows. These issues are addressed in the companion volume (OECD forthcoming).

As Chapter III explains, the economic values attached to ecosystem functions are derived from the preferences that individuals have for those functions. In turn, the preferences are measured through the notion of willingness to pay (WTP) to secure or retain those functions and services. One clear advantage of this approach is that the benefits of ecosystem functions are expressed in the same units, money, as the benefits of the land conversion process that threatens biodiversity. Direct comparisons can therefore be made, whereas other value systems (see Chapter III) do not have this advantage. Moreover, the kinds of economic value discussed later in Chapter VI can be divided into use and non-use values, i.e. WTP based on the uses made of ecosystems, however indirect, and WTP based on people's concern simply to conserve systems or component parts of systems (such as specific species) regardless of any use made. The resulting sum of use and non-use values ('total economic value') then describes the economic value of the ecosystems.

Resilience

The second link between diversity and ecosystems is the effect of diversity on resilience. The diversity-resilience link has consequences for the way we think about value of biodiversity. As Perrings (1995) notes:

> 'the link that is now being emphasised between functional diversity and ecological resilience…changes our perception of the effectiveness with which the problem [biodiversity loss] may be addressed at the global level. This is because it changes both the time path and the geographical distribution of the benefits of biodiversity conservation. The effectiveness of biodiversity conservation at the global level is a function of the geographical distribution of benefits' (p.69).

Perrings' view is that the main consequences of diversity loss lie in the loss of resilience that in turn will show up mainly in local losses rather than global losses. Loss of resilience will affect other users of a given ecosystem. Since negotiations are difficult when the number of parties are large, the localised consequences should make effective action more likely.

Moreover, once the focus is on the way ecosystems may change in the presence of stresses and shocks, it is important to note that the processes of change may not be 'linear'. For example, a modest change may result in some dramatic effect rather than an equally modest one. The process of change is marked by *discontinuities* and potential *irreversibilities*. Equally, some major changes may have little effect on the system.[8] Resilience measures the degree of shock or stress than the system can absorb before moving from one state to another very different one. Diversity, it is argued, stimulates resilience perhaps because individual species threatened or affected by change can have their roles taken over by other species in the same system. The smaller the array of species the less chance there is of this substitution process taking place.

From an economic standpoint, the issue is one of identifying and measuring this insurance value. Unfortunately, neither is easy. Identifying how

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8 The explanation here lies in where the ecosystem is relative to equilibrium states. There may be multiple equilibria some of them being stable and some unstable. If a system is close to an unstable equilibrium a relatively small stress or change could result in a catastrophic reaction (Perrings, 1995).

close a system might be to collapse, of some or all functions, is extremely difficult. Yet one would expect willingness to pay to avoid that collapse to be related to the chances that the collapse will occur[9]. If the probabilities are known, the value sought is then the premium that would be paid to conserve resilience. Suggestions include the entire cost of managing non-resilient systems, since these costs would be avoided if more diverse and therefore more resilient systems are adopted. In the agricultural context, for example, this would make the premium equal to the entire costs of ensuring that intensive agriculture is maintained, including such things as fertiliser and pesticide costs. Inverting the process, it could be argued that the premium is approximated by the cost of all the losses incurred by maintaining a resilient system. If, as was suggested earlier, diverse/resilient systems are lower productivity systems, then the loss of productivity from maintaining a resilient system might be thought of as the economic value of resilience, i.e. as the resources that have to be sacrificed to maintain diversity.

Overall then, the economic value of biodiversity loss comprises two major components:

a. the use and non-use values associated with loss of ecosystem function; and

b. the premium associated with the loss of ecosystem resilience to change.

The former notion is more oriented to biological resources but includes a strong element of the value of diversity, whereas the latter is more heavily oriented to the value of diversity. In this way, the notion of the economic value of diversity could encompass the two interpretations of the meaning of biological diversity: as biological resource and as diversity per se. In one sense, valuation happens implicitly if not explicitly. All choices imply that those rejected have less value than those accepted. Correctly used, valuation can help make a powerful argument for conservation of biological resources and biodiversity. However, economic (non-market) valuation cannot guarantee the precision that conservation biologists, taxonomists and ecologists might seek. It is very much a surrogate approach for delivering the value that lies in the diversity of complex ecosystems. The economic approach is anthropocentric and tends to assign value to elements of biodiversity that are known and understood. These value criteria may not always match the

9 The analogy would be that individuals should be willing to pay more to reduce risks to their lives the bigger the current risk is. Empirical evidence does not, however, offer much by way of support for this presumption.

elements that are actually life-sustaining. At the ecosystem level in particular the inherent nature of value is poorly understood, especially how interconnected systems work and the nature of externalities arising from perturbation to complex systems. Extending the boundaries of current valuation methodology requires a better understanding of ecosystems and their functions, especially the identification of those functions that are irreplaceable system links. Some understanding of marginal versus non-marginal system impacts is also vital.

2.9 Non-economic values

Economic values reflect individuals' preferences for or against the object being valued. Thus, the economic value of biological diversity could be small if individuals reveal a low preference for its conservation. Many of these preferences will be for activities associated with the actual use of biological resources and diversity: eco-tourism, indirect consumption through viewing wildlife films etc. But there will be many indirect uses where a conscious act of valuation is not practised: the role of ecosystems in maintaining a clean environment, for example, has an economic value which is not revealed directly but indirectly through the value of that clean environment. As noted earlier, diversity is also information and a pool of resources that act as inputs to modern agriculture and medicine. Hence, the economic value of biodiversity could be extensive, ranging as it does across the many functions of biodiversity.

There is a view that economic approaches risk omitting whole categories of value residing in biodiversity (see van Ierland *et al.*, 1998). Two sources of omission occur. First, there may be economic values that are hard or even impossible to identify. If, for example, diversity is critical to entire ecological life-support systems, then diversity has a use-value which may be hard to measure. A second possible source of omission arises from religious or spiritual concern to protect diversity, values that may be formalised in organised viewpoints such as religious movements, or in more scientific concerns, as with the Gaia movement (Lovelock, 1979). While much of the conservation literature debates the economic and the non-economic approaches, the more relevant issue is what questions the different approaches are best suited to answering. If the issue is awareness raising, all approaches are probably relevant: arguments that appeal to some people will not appeal to others. If the issue is one of choosing between losses of biodiversity in order to make gains in other areas, the economic approach is potentially very useful, but so is an informed participatory debate in which economic aspects are one ingredient. Some values, however, do not lend themselves easily to making choices. Notions of 'intrinsic value', 'primary value' and 'spiritual value', for example, would all be relevant to awareness raising, but may not assist in making choices that necessarily involve sacrifices. In some contexts the primacy of these non economic values will simply make the consideration of economic opportunity costs appear irrelevant.

It is important to understand that there are different value systems relating to choices about conservation and sustainable use of biodiversity. A distinction between intrinsic and anthropocentric approaches can frequently be detected in debates about resource allocation. Those who believe that biodiversity has an intrinsic value - a value 'in itself' and independent of human valuation - will want to argue that it cannot be traded against notions of resource cost because: (a) intrinsic value cannot be measured; and, (b) cost is an anthropocentric concept which cannot be compared to intrinsic value. Those who accept that all decisions about conservation involve costs may prefer to see the benefits of conservation brought directly into comparison with those costs; a view that underlies the economic approach.

Perhaps the most useful point that can be made is that the different approaches need to be articulated clearly and then applied to the questions that are likely to be relevant in terms of policy decisions.

III. VALUES AND DECISION-MAKING

3.1 A typology of values

Philosophers dispute the meaning of the word ‘value’ and whether, however defined, value resides ‘in’ the objects of interest (objective value) or is conferred upon the object by the entity engaging in the act of valuation (subjective value). Any attempt to classify ‘types’ of value will therefore be tendentious, but the following broad categories are often found to be helpful:

a. Instrumental, or functional value.

b. Aesthetic value.

c. Moral value or ‘goodness’.

Instrumental value derives from some objective function, goal or purpose that is being sought. As an example, economic value relates to the goal of maximising human well-being (or welfare, or utility), where well-being has a particular connotation, namely that someone’s well-being is said to be higher in situation A than situation B if they prefer A to B. It is immediately obvious that economic value is *anthropocentric* and it is *preference based.* There can be no dispute that biodiversity has economic value. But it may not be the only value that it possesses.

Aesthetic value is a non-instrumental value, even though it is a value expressed by human beings. It is usually regarded as being non instrumental because beauty - the concept most widely referred to in speaking of aesthetic value - is regarded as an end in itself, not as a means to some other end. Arguably, beauty has an instrumental element because appreciating beauty affords pleasure and a sense of well-being. As with other notions of value, philosophers debate whether beauty is ‘in’ the object itself. If it is, then this ‘real’ characteristic of the object interacts in some way with the person perceiving the object so as to provide them with the sensation of beauty. The philosopher G.E. Moore argued that beauty is an objective value, surmising that what is beautiful or ugly would still be so even if all humans did not exist. Others argue that the aesthetic value of something is determined by the person engaged in the act of value: what appears beautiful to some may not appear beautiful to others. Moore's 'thought experiment' is, they argue, meaningless since value cannot exist without a valuer. That there exists some broad consensus across most people that certain things have beauty suggests either that many people simply share the same 'taste' or that there is a property of objects that gives rise to expressions of beauty. Perhaps both are relevant: aesthetic value is 'in' objects but only exists because there is a 'valuer'. Biodiversity in the sense of diversity may not be the subject of aesthetic value, but many of its components clearly are. In effect, there are attempts in many countries to support and restore agricultural landscapes and agrobiodiversity for reasons of aesthetics (OECD, 2001 d).

Moral value is also often non-instrumental, perhaps more clearly so than aesthetic value. The 'goodness' of an act could be defined in terms of the well-being the act confers, but is usually more widely concerned with acts being just, right in themselves or simply 'good'. Philosophers debate the source of moral value: to say 'X is good' may mean that the person making the statement simply likes X, that X can be rationally derived as a good thing, that goodness resides in X like an objective quality, or that X is good because a body of religious doctrine says it is good. But moral value can co-exist with instrumental value if what is moral or right is that which achieves some objective, such as human well-being. Many people feel that the loss of biodiversity is a moral 'bad', something that simply is 'not right'. Again, philosophers debate whether this moral value resides in the object of interest or whether it is conferred on the object by the valuer. If it is objective, residing 'in' the object, then it will exist regardless of whether humans exist as the valuers. The terminology for such objective values usually involves notions of *intrinsic* or *inherent* value. If moral value is subjective, on the other hand, then moral value is whatever the valuer thinks it is. The subjective-objective value debate is a long one in the history of philosophy. Thus, Immanuel Kant regarded human beings as being 'ends', i.e. as having intrinsic value.

The distinctions between instrumental, aesthetic and moral value are not precise. Consider economic value again. It is one form of instrumental value: something is valuable if it contributes to the goal of maximising well-being, and has negative value if it detracts from this goal. Exactly what constitutes this well-being, what it 'contains', has traditionally not been the concern of economists, although there is a growing interest in this issue. This interest usually shows in terms of discussions about the *motivations* for a preference: why that preference is held. Preferences may be for a variety of motives: self-interest, concern for the immediate members of one's family, concern for other human beings, other sentient beings, any other life form, the well-being of the planet, and so on. Some motivations may therefore be derived from a concern that the object of value has a value 'in itself', i.e. an *intrinsic value*. Values that acknowledge such motivations are said to be *anthropogenic*. Anthropogenic values are necessarily 'of' people but may include concerns 'for' the object of value. On the other hand, *anthropocentric* values necessarily confer value only because of the effects of the object on humans. There is no element of intrinsic value.

Instrumental values will clearly vary according to the objective function in question. An objective function like 'maximising human well-being' is deliberately general because it allows individuals to have very different motivations for their preferences. Individual A may want to conserve biodiversity because it gives him or her pleasure; B might want to conserve it for future generations, and C might want to conserve it because he or she holds that biodiversity is an end in itself. This mix of motives does not make it impossible to add up the resulting preferences since, whatever the motive, it may be revealed through willingness to pay.

While instrumental values tend to be capable of measurement on a scale, aesthetic and moral values tend to take on the 'zero-one' characteristic. Something is either beautiful or right, or it is not, although everyday language speaks of things being 'very' beautiful as well as beautiful. Others would argue that things may be morally right but that deviations from what is right may not matter too much in some contexts. The zero-one characteristics are very important in the context of practical policy, as will be explored below.

Because economic value is most widely contrasted with moral value, it is helpful to classify moral value a little further. Figure 3.1 suggests a possible decomposition.

Figure 3.1 A typology of moral values for biodiversity

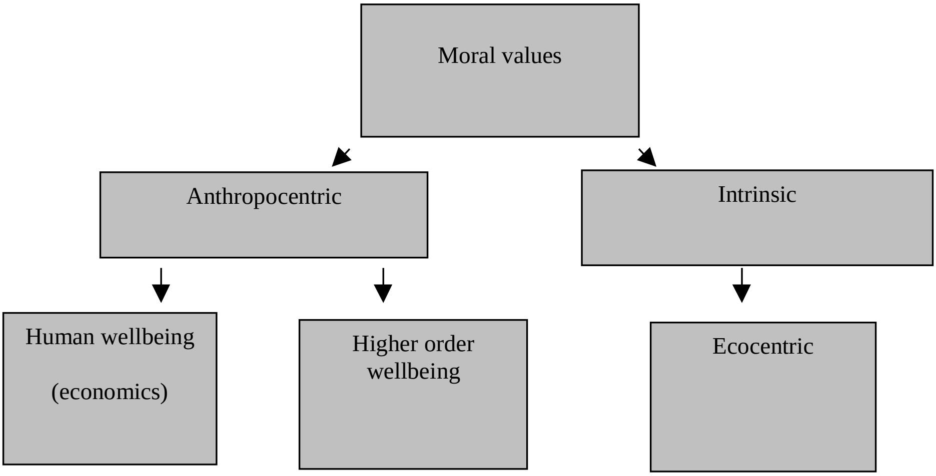

Figure 3.1 is necessarily simplistic, but suggests that moral value is either anthropocentric or concerned with intrinsic value (but there is a ‘mixed category’ that involves both, as discussed under instrumental value above). Anthropocentric value would justify conserving biodiversity because of the contribution it makes to human well-being, or because of the contribution it makes to some ‘higher order’ notion of well-being. The difference is essentially that the former does not look too much into the ‘content’ of human well-being: whatever individuals prefer defines that well-being. The latter is concerned with the content because it regards notions like ‘improvement of the self’ or a ‘richer life’ as circumscribing the kinds of things that are to count in human well-being. Biodiversity is often seen to be very important in this latter context because appreciation of the complexity and wonder of life forms (e.g. a wilderness) is thought to ‘transform’ individuals into better human beings or to contribute to a better, more cohesive society. This ‘higher order’ approach would be associated with writers such as Norton (1986) and Sagoff (1988) deriving from earlier writing such as those of John Stuart Mill. Clearly, there can be a moral debate about the anthropocentric categories of moral value: some may feel individuals are sovereign and that ‘higher order’ preferences sounds like a cultural elite imposing values on others. Equally, no society functions by letting individuals have unconstrained preferences, or, if it does, it does not organise resource allocation to meet those preferences.

Approaches to biodiversity conservation based on ‘ecocentric’ ideals tend to confer intrinsic value on the objects in question. There are varieties of ecocentric views. Some are restricted to conferring moral value on sentient life forms only, others are narrowly to animals only (animal rights), and some to 'systems' of life forms, e.g. ecosystems. In the latter case, most popularly associated with Aldo Leopold (1949) what matters is the health of the ecosystem generally, and the rights and duties of individuals derive from that goal. Each individual has to behave so as to conserve that ecosystem's health. This is similarly how some followers of the 'Gaian' ethic would articulate their views: their rights and duties are defined by what has to be done to conserve planetary survival or sustainability and health.

This very brief description of different approaches to value systems in the context of biodiversity is necessarily impressionistic. The essential distinction is between notions of instrumental and intrinsic value. Instrumental values are conferred by the valuer for some human purpose. Intrinsic values are like objective features of the object in question: 'colour' or 'shape', for example. But this simple division between instrumental and intrinsic obscures the possibility that subjective value may involve ascribing instrumental value or intrinsic value. The subjectivist approach effectively declares that there are no 'objective' values and if intrinsic value is an objective feature, then subjectivism is not consistent with intrinsic value. But others argue that while there can be no notion of value independent of a valuer (i.e. they believe in subjectivism) the value that is conferred on objects could be instrumental or intrinsic (Beckerman and Pasek, 2001).

3.2 Debates about value systems

A test of the relevance of the different value approaches is whether they can be applied, at least in principle, to practical issue of biodiversity conservation. If they cannot be applied, it does not make the approach 'wrong' in any intellectual sense. It may simply be infeasible, and feasibility naturally varies with the institutional and other conditions within which decisions have to be made. Just a few issues are selected for discussion. The reality of decision-making means that many decisions will be very complex.

Intrinsic vs instrumental values

In principle, any value system can be used to decide whether biological diversity should be conserved. An instrumental approach would normally involve some participatory process whereby individuals agreed on the objective function in question. Conservation would then be justified according to whether it met the objective function or not. The issue, as with all approaches, that makes things more difficult is the requirement that conservation be allocated real resources. The notion of *opportunity cost* refers to the fact that the allocation of resources to biodiversity conservation necessarily means those resources cannot be allocated to something else. From an economic perspective, the money value of the resources allocated to conservation approximates the benefit that is sacrificed for conservation. Hence, for the instrumental value rule to be obeyed, it must be the case that the benefits (positive changes in human well-being) from conservation must exceed the costs of conservation (the well-being foregone). In essence, that is the resource allocation rule that would be used in economics.

On the intrinsic value approach, cost remains relevant, although it is unusual in an environmental ethics debate to find it discussed. For just as intrinsic value may reside in biodiversity, so it may reside in the benefit that is sacrificed by allocating resources to conservation. While an intrinsic value rule looks as if it is 'absolute' - if something is good, or right, it has to be done - in fact it may well involve a trade-off between different kinds of 'goodness'. The 'right' to a clean environment may conflict with the 'right' to an old age pension or healthcare. The instrumental approach has a scale of desirability and it is this that permits it to compare gains and sacrifices. Intrinsic value approaches do not have such scales, although some advocates of the intrinsic value approach would argue that biodiversity has 'higher order' value than other objects in which intrinsic value resides. These kinds of comparisons clearly become very difficult in contexts where, say, biodiversity conservation conflicts with the well-being of very poor populations, as can be the case for tropical forest conservation, avoiding rangeland degradation, etc. But instrumental approaches may also face problems in such contexts. Although the well-being of poor people would count in the instrumental approach, use of measures such as willingness to pay could bias outcomes against the poor and be in favour of the rich. Instrumental approaches may therefore need to be tempered with concerns for notions of justice.

Instrumental vs higher order instrumental values

The idea that only some preferences 'count' and others do not is widespread. Governments world-wide regulate what is allowed to dictate policy and what is not. Individuals might be thought to be ill informed or fulfilment of individuals' preferences might conflict with some broader goal of social harmony. Few societies exercise a totally free choice over whether or not to attend school, whether or not to commit crimes, and so on. Hence restraining choices is one of the features of the modern state. There is a perpetual debate about how far that process of intervention in 'free choice' should go. To the economist, free choice is a working hypothesis and restrictions are justified on the basis of the costs that free choices may impose on others. Few economists would defend having no speed limits on roads. Few would defend the wholesale destruction of biodiversity. In both cases, the notion of correcting free choice to account for ‘externalities’ would be a powerful justification for restricting choice. The obvious problem is that, once restriction of choice is permitted, it invites some groups to dictate to the rest of the population what is good for them. The appreciation of many cultural events, for example, is often confined to a limited group in society. Whether that group’s interests should be met with public resources paid for by everyone is then a debatable matter. Those who believe in ‘transformative value’ - the notion that something should be provided because people will grow to appreciate it and will be transformed in some way by the experience - would argue that public resources should indeed support such goods.

The zero-one dilemma

The fact that instrumental value systems can have scales of desirability has already been noted. Notably with respect to the economic approach, this permits a comparison with what is sacrificed by making the decision. The non-instrumental approach would effectively amount to saying that conservation of this wetland or that forest is intrinsically good. The trade-off with cost might then be left to the political process without any formal calculus of gains and losses being involved.

But this zero-one feature of intrinsic value raises a further problem. If biodiversity is intrinsically good, it would seem that *all* biodiversity is intrinsically good. It is hard to see how the moral value attached to one forest is any different to that attached to another forest with the same or similar biodiverse features. In the extreme, the intrinsic value approach amounts to saying that all biodiversity has to be conserved. Within the ecocentric paradigm, some environmentalists circumscribe what it is that they regard as worth conserving: just animals (not all life forms or sentient life forms), for example. This could be thought of as a mechanism for avoiding some of the implications of the zero-one characteristic; that is, the problem becomes more manageable by restricting what counts. Thus, if the zero-one feature of the intrinsic value rule could be relaxed in some way, it is easy to see that the problem becomes more manageable. It is not in fact possible to conserve all biodiversity, so various practical rules could be envisaged. Perhaps the rule would be to conserve whatever can be conserved. Since biodiversity is strongly linked to land area, perhaps the most diverse areas should be the subject of conservation first. But problems remain because the intrinsic value approach does require that any budget constraint be relaxed until all biodiversity is conserved, itself an impossibility. In short, the intrinsic value approaches do not permit trade-offs and many would argue that it is difficult to see how trade-offs can be avoided. Of course, even though trade-offs are necessary, it does not follow that the economic approach is the only way of making those trade-offs. Political processes might be used instead.

3.3 Can conservation policy be value-free?

The concept of value pervades the entire issue of public policy choice, just as it pervades private choice. Policy on biological diversity is no exception. For some, biodiversity - the ‘web of life’ - is so important that its value transcends the value of other things. Without biological diversity, there can be no human existence. Hence the idea of ‘trading’ biodiversity against other things is not acceptable to some people. At the other extreme are those who argue that the value of biodiversity derives only from the some human goal - meeting human demands generally, or the demands of some specially defined group of humans. The context in which value debates take place, however, cannot be one in which everything can be preserved, nor one in which all human goals can be met. Hence, there has to be a choice.

Most people accept that difficult choices have to be made, but even with that acceptance, very fierce debates take place about how much diversity to save and which diverse areas of the planet should be conserved first. Environmental philosophers who accept that there is a limited ‘conservation budget’ opt for some form of *cost-effectiveness* criterion. Thus, Norton (1987) argues that a cost-effectiveness criterion could be adopted based solely on what he calls *formal* criteria. Formal criteria involve rankings of species that do not have to refer to characteristics of the species in question (e.g. their ‘attractiveness’ or ‘importance’). An index of species richness would be a formal ranking, but an index of richness where each species was weighted by some indicator of importance or its own characteristics (such as longevity) would be a *substantive* criterion. The essential difference, Norton argues, is that formal criteria involve no controversial value judgements, whereas substantive criteria do. A ranking of species by richness and endangerment would similarly be formal, not substantive, assuming that everyone can agree on what the indicators of threat are. A process of prioritising species conservation could therefore be ‘value-free’. Norton suggests that rankings would remain value-free even if they included taxonomic status, based on phylogenetic trees (see Chapter II). Human values would not enter the analysis because such measures of species distinctiveness are ‘scientific’. The goal of conservation would be to maintain the most diverse gene pool possible, which Norton sees as an end in itself, rather than as a means to an end such as human survival or human well-being. Norton regards this approach as being based solely on *'ecological value'*, a scientific measure of value.

Norton's approach can be construed as a cost-effectiveness approach. The goal could be restated as one of maximising the expected value of diversity, where 'value' refers to ecological value. The term 'expected value' denotes the probability-weighted value of species, where the probabilities in question are those of extinction of a given species. Such a goal is strongly identified in the economics literature as one of maximising *option value* (see Chapter IV**)** which refers to the value that a species might have in the future. Individuals would be willing to allocate resources to conserve that species not because they make 'use' of it now but because they (and future generations) may make use of it later on. The term 'use' here has to be interpreted widely, e.g. the contribution of species diversity to ecosystem resilience would be included in future value. It seems more likely that a set of species containing more *genetic distance* than another set will have a higher option value. Survival probabilities are also maximised because species that are genetically similar are likely to have similar resistance to threats.

It can be seen that devising a conservation strategy that is 'value-free' is feasible but not necessarily very helpful. In part, the drive to find a value-free approach may reflect a misunderstanding about the nature of the values that are likely to be relevant in a value-laden approach. It seems clear that diversity affects outcomes, such as resilience, which are valued by individuals. Nonetheless, it is true that the expected value of diversity approach does not necessarily require the kinds of valuations involving monetary measures or political processes.

If the arguments of environmental philosophers and economists are not so far apart, it does seem clear that both are very far away from much of the current conservation practice. Currently, some resources do get allocated to 'exotic' species conservation (giant pandas, for example) without any real consideration of the diversity issue. In other cases, such as the US Endangered Species Act, not only is the issue of ecological value (genetic 'distance') ignored, but so is the issue of the cost of protection[10].

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[10] In the 'snail darter' case, Tennessee valley Authority vs. Hill, 437, US 153, 184 (1978), the Supreme Court rules that 'The plain intent of Congress in enacting [the Act] was to halt and reverse the trend towards species extinction, whatever the cost'.

3.4 The goals-alternatives matrix

All decisions involve stating a goal or set of goals, and considering the different alternatives for achieving those goals. In the case of biodiversity, the goals may include conserving a given species, conserving habitats, conserving information in ex-situ gene banks, setting priorities for areas or ecosystems to be conserved, and so on. But there will also be a concern about cost since, as noted in Chapter II, cost implies forgone goals in other areas of biodiversity conservation or in some other area of policy. Cost-minimisation may therefore be seen as a goal, or cost may be viewed as a constraint on securing the biodiversity goals. Other goals might include employment creation or other environmental benefits besides biodiversity conservation. The means of achieving these goals may include giving certain areas protected status, subsidising activities that benefit biodiversity, encouraging eco-tourism, penalising activities that harm biodiversity, involving local communities in the protection activity so as to provide them with 'ownership' of the biodiversity, and so on. A *goals-alternatives matrix*, such as that shown in Figure 3.2, sets out the hypothetical relationships between goals and the means of achieving those goals.

Figure 3.2 Goals-alternatives matrix

Goals↓ Alternatives→	Establish protected area	Community involvement	Financial incentives, tax harmful activities	Weights
Improve an indicator of biodiversity	+ 5%	+ 1%	+ 8%	3
Employment	- 1%	+ 2%	- 1%	1.5
Other environmental benefits	+ 2%	0%	+ 7%	1
Cost	$1m	$0.4m	$2m	-
Weighted score of benefits	15.5	6.0	29.5	-
'Cost-effectiveness' indicator	15.5	15.0	14.75	-

The matrix is to be read as follows. There are three 'goals' in biodiversity conservation: increasing biodiversity, as measured by some selected indicator of diversity, increasing employment and securing some other set of environmental benefits. These goals are not equally important, so weights are applied to each of them. Taking 'other' environmental benefits as a numeraire, conserving biodiversity is three times as important and hence has a weight of 3, and employment is 1.5 times as important and hence has a weight of 1.5. There are three different ways of securing the goals: establishing a protected area, adopting a community involvement scheme, or some tax on harmful activity. Each option or 'instrument' is evaluated according to the extent to which it secures the relevant goals. Thus, a protected area is estimated to improve the biodiversity index by 5% but to reduce employment (say in the local area) by 1%. The weighted scores of benefits are then obtained by summing the achievement scores (the percentages) weighted by the importance weights. For example, the protected area option scores (+5x3)-(1x1.5)+(2x1) = 15.5. Finally, consideration is given to cost. The weighted scores can then be divided by cost to secure a cost-effectiveness indicator. This shows that the protected area scores the highest.

The matrix reveals that there are multiple goals and that the different approaches to achieving them have different costs. The issue of deciding which of the methods to use (or, additionally, how they might be combined) becomes one of comparing costs and the extent to which the goals are achieved. Taking biodiversity conservation alone, the most effective measure is the tax on harmful activities but it is also the most expensive measure. In terms of cost-effectiveness, the 'best' option is the protected area. Both the tax and the protected area option actually reduce employment. Only the community options involves an increase in employment. Thus, the matrix reveals that, on the basis of the information provided, no alternative is clearly superior to the others. Only the weighted score approach produces a ranking.

The fact that outcomes are often sensitive to the chosen weights suggests that the focus of attention should be on the justification of the weights. The options for finding 'rational' weights are fairly limited: public opinion would be one, the use of experts would be another. The economic approach would effectively be a combination of both since the weights would be prices, i.e. willingness to pay. In turn willingness to pay will reflect people's preferences for or against the various outcomes, but that willingness to pay may also be influenced by expert opinion.

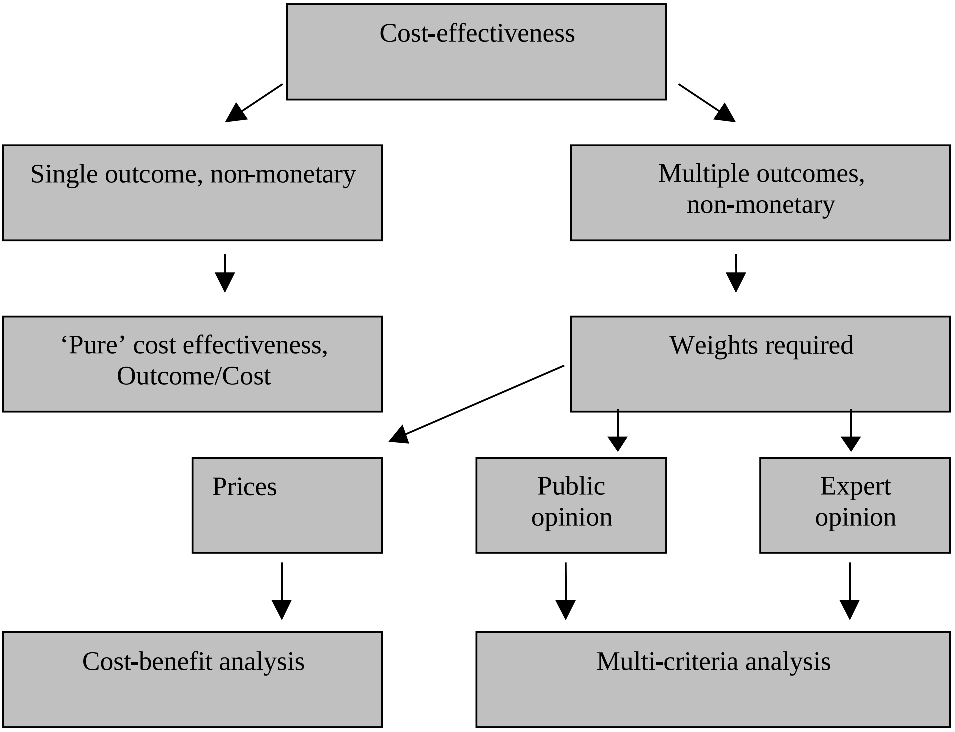
3.5 Weighting in alternative decision-making procedures

The matrix approach in Section 3.4 focused on the way in which weights affect the choice of policy measure. The initial selection of impacts to be considered provides the first indication of the role of value in decision-making. Selecting these impacts but omitting others means that the omitted impacts have zero value. Only those included in the list have value. Next, the weights confer *relative values* on the impacts, i.e. they indicate that some impacts are more important than others. More than this, they indicate the *rate* at which those relative values are to be traded off.

The different ‘value procedures’ can now be illustrated using the matrix in Section 3.4. But an important issue before looking at these value procedures is to determine what questions they can answer. The relevant questions are: (a) which of the alternatives constitutes the ‘best’ choice, and (b) should *any* of the alternatives be chosen. The second question is often omitted in practical decision-making, but its relevance lies in the fact that most decision-making procedures cannot answer that question. The reason is simply that both gains and losses, benefits and costs, must be in the same units for the question to be answerable. Figure 3.2 shows that the protected area is the best option of the three, but it is not possible to say that any of the three options should be chosen. Perhaps the best option is to do nothing. Doing nothing would cost nothing, but biodiversity would decline. A cost-effectiveness indicator would have no meaning because there would be no cost.

In practice, decisions are usually constrained so that something has to be done. Provided the options are well defined and provided they constitute the whole set of feasible options, a cost-effectiveness approach may suffice. Thus, a good first step for making decisions is to adopt cost-effectiveness. But cost-effectiveness will suffice if there is only a single ‘outcome’ (say biodiversity gain) and the choices relate directly to that outcome. Cost-effectiveness becomes more complex when there are multiple outcomes since the outcomes have to be weighted. Weighting may involve three dimensions: weighting the outcomes now, weighting the different time periods when the outcomes occur and weighting where the outcomes occur. The first, was discussed above. The second, is the issue of how future gains and losses should be discounted. The third, is of concern when we consider that biodiversity is both a local and a global public good. Focusing on the issue of current weighting, such weights can come from public opinion, expert opinion or the weights can be prices (willingness to pay). If the weights are prices, cost-effectiveness is formally transposed into *cost-benefit analysis*. If the weights are not in price form, cost-effectiveness becomes *multi-criteria analysis*. These relationships are shown in Figure 3.3.

Figure 3.3 Cost-effectiveness approaches



In the multi-criteria context, the weights are derived from individuals' preferences (public opinion or economic valuation) or from expert opinion. The *sources* of these opinions, i.e. the factors that determine what those opinions are, will be made up of numerous factors, including the instrumental and intrinsic value concepts previously introduced. It can be expected that individuals having strong views about intrinsic values are more likely to express strong opinions for, or place a higher willingness to pay on, biodiversity. The same will be true for experts. Two potential differences between expert and public opinion will be in the level of information possessed by the groups: in general, experts will be better informed, and they then will have more experience of biodiversity. Experts are more likely to have an understanding of the notion of *diversity*, although both groups can be expected to have an understanding of biodiversity as *biological resources*.

These issues of experience and information could be thought of as favouring the use of expert opinion in multi-criteria contexts. If the multiple goals all relate to biodiversity, this is likely to be the case. But cautions about this conclusion arise from the fact that cost is also a goal (or constraint) and there need be no reason to suppose that experts are any better at comprehending opportunity cost than the general public. It may also be the case that some of the goals relate to issues on which the experts have no expertise, e.g. employment impacts. Finally, even if experts are better informed and more experienced in the 'good' in question, there are reasons of good governance for consulting the general public as would be the norm in a democratic society.

The different philosophical approaches to 'value' can therefore be expected to influence the weights applied by experts and the general public to the multiple goals that may be present. But some of the philosophical debate relating to the environment in general, and biodiversity in particular, concerns the possibility that individuals may be unwilling to 'trade-off' gains and losses. For example, in terms of Figure 3.2, individuals might opt for the alternative that produces the greatest improvement in the index of biodiversity, regardless of cost and regardless of any other benefits or costs associated with that option. Such individuals are said to have 'lexical' or 'lexicographic' rankings: whatever benefits biodiversity most will always be ranked first. Effectively, 'lexicality' denies trade-offs. The notion is consistent with everyday sayings such as the good being 'beyond price' or 'priceless'.[11]

Valuation approaches that utilise 'stated preferences', i.e. questionnaire approaches to eliciting values, are well suited to discovering whether lexical preferences exist. If they do, respondents to the questionnaire will generally give 'protest' responses to questions that call for the respondent to provide a weight or price. Some questionnaires have claimed that a percentage of respondents give such responses, but the issue is debated in the literature and it seems no firm conclusion can be reached at this stage.

3.6 Multi-criteria approaches

Problems encountered in attempting to place reliable values on all environmental impacts have led some analysts to conclude that any single criterion of value is inappropriate, and that the multidimensional nature of environmental change requires several criteria to be assessed in ranking the options. For example, ecosystem perturbation may result in the loss of a wild species associated with a market value. It may also lead to the damage of a suite of natural functions that cannot be represented by analogous economic values.

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11 But not with the notion of an infinite price which is self-evidently meaningless.

However, ecological specialists may find it possible to rank these functions in terms of their contribution to the health of the whole ecosystem. As noted in the previous section, multi-criteria analysis (MCA) (and the closely related concept of multi-attribute analysis) is an alternative form of decision-making that explicitly addresses the multiple objectives in decision-making. In environmental terms MCA attempts to allow monetary and non-monetary units to be assessed side by side. For example, a project or policy's net economic benefits may be one important criterion that is to be considered alongside a number of important but incommensurate physical indicators for which monetary values are not derived. A list of biological indicators of ecosystem health may be one such criterion.

There are several reasons why MCA may be preferred by decision-makers themselves. Environmental changes are sometimes perceived as being too complex and multidimensional to be reduced to single criteria such as economic efficiency. Second, the concept of economic efficiency itself can seem too abstract for decision-makers. Frequently it is the case that specific elements or an environmental problem (e.g. water pollution) have precipitated the need for a project. Accordingly, a non-economist decision-maker may focus on the options to solve that problem irrespective of the efficiency criterion. This was the issue raised in the discussion of cost-effectiveness, namely that only a criterion in which gains and losses are in the same units can determine whether a decision is efficient in the overall sense. Third, the absence of valuation information may necessitate an alternative weighting approach.

Although there are several variants on the MCA process for ranking options, the basic steps in conducting the variants are similar:

a. specify objectives and project alternatives for meeting objectives;

b. select criteria for assessing or ranking alternatives;

c. specify the selection system to be used as the basis for making decisions, i.e. the relative priorities or weights to be attached to the criteria selected in (b);

d. identify global performance of alternatives using some method to combine the weights into a final score for each alternative (see Nijkamp *et al.,* 1990).

The process can be made more or less elaborate depending on the level of information available to the decision-maker and the modelling sophistication. Thus for example, stages b and c can be based on complex models, which identify environmental impacts and model the relationship between impacts and environmental quality (e.g. water), which then are assigned a weighting or scoring system according to the expert view of the importance of that single attribute (water quality impact). It is typical to normalise the impact – quality relationship such that the latter has a 0–1 range. Table 3.3 represents a typical valuation matrix using similar information.

Table 3.3 Valuation Matrix

Environmental Criteria	Alternative Projects			
	A	B	C	D
1 Water quality	0.8	0.9	0.6	0.6
2 Soil erosion	0.5	0.7	0.6	0.7
3 Air pollution	0.6	0.9	1	1
4 Tree species	0.8	0.8	1	1
5 Mammal species	1	1	0.7	0.6

In Table 3.3 five environmental criteria have been identified and scored using normalised impacts (0 = very bad, 1 = very good). They are applied to four different options. Suppose that these options have approximately equal economic benefits such that the economic criterion can be removed from consideration. The economic benefits are assumed to exclude the criteria shown, i.e. no economic values have been derived for water quality etc. The next question is how the combined score for an option should be compared to that of alternatives. The most rudimentary method to determine the favoured option is to check whether options clearly dominate or are dominated by alternatives using all criteria. If this is not the case then the expert opinion can be called upon to provide a weight or score to each criterion according to its perceived importance in this case (e.g. is water quality more important than air pollution?). These weights will be additively equal to 1. There are numerous methods for deriving weights and for deriving the final ranking of options using the combined impact and weight information[12]. These decisions suggest that the MCA process is necessarily subjective. This is for the primary objectives of the investment or policy, but more controversially for the process where weights are set for the criteria identified as relevant to the decision. Both steps can be determined by stakeholder or expert Delphi consultation processes, which are group exercises for determining scientific or social consensus about priorities.

[12] Weighting technique include Delphi Methods, Paired Comparison and Opposed Pair Methods. Ranking methods include weighted worst score methods, concordant and discordant matrices and dominance approaches.

Box 3.1 Using multi-criteria approaches in hydroelectric power planning in Norway

The development of watercourses for hydroelectric installations presents a number of apparently incommensurate environmental, economic and social trade-offs. Environmental impacts include altered flow levels and habitat disturbance in rivers and in surrounding areas. While Norway may be particularly well endowed with pristine water resources, the National Master Plan for Water Resources recognises the need to prioritise sites for hydroelectric developments taking all impacts into account. The plan is developed as a strategic tool in the national Ministry of Environment, in conjunction with the Ministry of Petroleum and Energy and the Norwegian Water Resources and Energy Board. It develops a multi-criteria methodology for ranking groups of projects that are deemed most suitable for development, accounting for a number of screening stages.

For each project, an initial screening was performed, 16 user interest/ topics were ranked on a scale of –4 (serious negative impact) to + 4 (positive impact). Among the topics are those concerning nature conservation, wildlife and fish, reindeer husbandry, water supply and pollution protection, and outdoor recreation. After its inception in 1984, 542 project sites were scored on this basis plus a separate score for the cost-effectiveness of energy production associated with a power development at the site. A ranking matrix was produced consisting of 8 impact categories by 6 economic (cost-effective categories). The project sites were then fitted into one of 48 cells and sorting by a preference function gave a resulting ranking into 16 groups, where group 1 represented projects with good economic return and low impact and group 16 represented projects with bad economic potential and serious impacts. The priority grouping was then ranked according to the impacts on local economic development and project size. Those fitting into the lowest impact group and the lowest energy cost groups were then prioritised concerning development.

The process shows how MCA can be pragmatically adjusted. Criteria can be derived for separate stages of the analysis and economic and environmental elements feature at different stages of the ranking. The Plan is an example of how MCA can be used in a national setting.

MCA treats economic efficiency as just one criterion and proponents of the method suggest that the simultaneous consideration of attributes is a strength that allows the method to parallel policy trade-off more accurately. But there are arguments for supposing that economic efficiency is a ‘meta-criterion’ because it determines the size of the benefit secured. Other considerations may reduce the size of the benefit, thus leaving less resources to be allocated to non-efficiency criteria. One way to adjust for this problem is to give a higher weight to the economic efficiency criterion. Second, while MCA can use public opinion, it is invariably the case that it uses expert opinions. Experts may, as noted earlier, be better informed and more experienced, but they may not reflect public opinion. Third, it is not clear how time can be incorporated into MCA, other than by presenting options that arise in different time periods. Yet time is a critical feature of rational decision-making (See Chapter V). MCA does not suggest a consistent approach to inter-temporal resource allocation**.** Lastly, MCA is only a way of deciding between schemes. It does not tell us whether any of the options are actually worthwhile in the aggregate sense of being welfare improving. Box 3.1 provides an example of MCA use in Norway.

3.7 Costs, effectiveness and precaution

Cost-based approaches

The preceding sections indicate that the notions of benefit and cost are central to practical decision-making, but that many decision-contexts may not require that the benefits and costs be measured in the same units (money). Monetising benefits has advantages when the issue is one of deciding whether any action at all is worthwhile. Otherwise cost-effectiveness analysis will select the ‘right’ option given that a choice has to be made from a defined set of options. Cost-effectiveness approaches do, however, imply weights. Selecting an option that has the highest amount of biodiversity conserved for a given expenditure, implies that the money value of the conserved biodiversity exceeds the cost, or, put another way, that the money value of the benefit from conservation exceeds the money value of the benefits that could have been obtained by using the resources for some other purpose. In this case the monetisation is not explicit but implicit and it is in this sense that monetisation is unavoidable (Thomas, 1963).

Multi-criteria approaches require weights in order that the analysis can be made tractable. As long as the weights are in non-monetised form, then multi-criteria approaches are akin to cost-effectiveness.

Whether the approach to decision-making is based on cost-benefit analysis, multi-criteria analysis or ‘pure’ cost-effectiveness, the common element to all of them is that cost is important. The two bases for focusing on costs are (a) that conservation expenditures are often (though not always) public expenditures which, in turn, represent taxpayers’ income, and (b) that what is spent on conservation could have been spent on something else.

Moral approaches

Approaches to value based on moral value may help determine preferences, but if applied in an absolute sense tend not to be cost-based. The justification for ignoring cost in this case is that what is ‘right’ or ‘good’ cannot depend on where society is willing to allocate resources. The difficulty with this approach has already been noted, since cost is effectively the command over some other good which may also be the subject of a moral view.

Precautionary approaches

A third set of approaches stresses the fact that we do not know precisely what will be lost if biodiversity continues to diminish. If the benefits include life-support functions, then biodiversity loss could take on catastrophic proportions. The context is one of genuine uncertainty, i.e. one on which the scale of the effect is not known, and the probabilities are also not known. Such contexts suggest some form of precaution, i.e. making the balance of decisions in favour of conserving biodiversity rather than losing it.

The most widely known principle embodying this idea is the *precautionary principle*. The exact nature of the principle is unclear, however. It emphasises prevention rather than cure, and it implies a significant degree of *risk aversion*, especially to change that is irreversible. Waiting for better information is also widely regarded in the precautionary principle literature as not a reason for tolerating risks. In these formulations, cost plays no role. Risk should be avoided whatever the cost of doing so. In other formulations, the principle comes closer to a risk-benefit assessment, i.e. risks are reduced if the costs of reducing them are tolerable or acceptable.

The *safe minimum standards* approach shares some of the features of the precautionary principle in that the ‘burden of proof’ is shifted to those who wish to increase the risk, e.g. reduce biodiversity. The SMS principle then states that biodiversity should be conserved unless the cost of conservation is, in some sense, ‘too high’. Thus, the SMS principle is not quite a ‘cost-free’ notion since it does acknowledge the trade-off context. The implicit value judgement, however, is that biodiversity has very large values, even if they are not currently known. Along with strong risk aversion, the SMS approach would make risk increasing activities more difficult to accept.

3.8 Conclusions

This chapter provides a general framework within which alternative approaches to determining the value of biodiversity can be discussed. The philosophical debate centres on the basic distinction between subjective and objective values, i.e. whether value resides in biodiversity or is conferred on the object by the valuer. The debate continues and it seems fair to say that both views have their very strong supporters. The debate is not the same as that between instrumental and intrinsic value because some philosophers claim it is possible to deny objectivism, whilst retaining the idea that people confer value and that what they confer may be instrumental or intrinsic or both. This view accords with the findings of questionnaire approaches to value where individuals often do say they want biodiversity to be conserved even though they may personally not be aware of any use they make of it, now or in the future (so-called ‘existence’ value).

While the philosophical debate is extensive, complex and largely confined to academic publications, it does have direct relationships to the practical problem of how to make decisions in a world where resources are limited. The relationships are sometimes weak, however, as with discussions that ignore the finitude of resources and hence the central place that has to be occupied by the concept of opportunity cost. Any practical decision-making criterion has to account for the benefits that are sacrificed by biodiversity conservation. This involves either a formal procedure, such as cost-benefit analysis, or some political process.

All practical approaches must also address the issue of how much biodiversity is to be conserved, given the impossibility of conserving all of it. The economic approach gives a direct answer to this question (that level which maximises the net benefits from conservation). Other approaches give less direct and less quantifiable answers, but may still be operable, e.g. ‘as much as can be afforded’, ‘as much as is possible’, ‘as much as is contained in the most species rich areas’, ‘as much as possible unless the costs of conservation are, in some sense, ‘high’, and so on. But logical limitations remain: if biodiversity has intrinsic value and that value cannot be measured, then all biodiversity should be conserved. This latter view may square with the notion of lexical preferences. It may even be an absolutist goal, or what is effectively a negotiating stance in the sense that the presumption should always be in favour of conservation unless the gains from losing it are very large.

All approaches need to have some form of consensus. The economic approach is based on the idea that people have organised the provision of markets when goods and services are scarce. If biodiversity is scarce, it can also be treated as if there was a consensus about it being allocated according to willingness to pay and opportunity cost principles. But if individuals indicate that this is not how they want biodiversity conserved, then the economic approach would have problems. Arguably the political process supports the idea that individuals do not want their preferences to dictate outcomes without reservation: individuals accept 'mutual coercion, mutually agreed upon'. If so, perhaps market-type choices should not dictate how much biodiversity is conserved.

Some decision-making processes stress the uncertainty of decisions about biodiversity, i.e. the difficulty of knowing what may be being lost. The precautionary principle and safe minimum standards both imply that biodiversity has substantial value even if that value is not known in any precise form.

Finally, policy needs to address the fundamental causes of biodiversity decline. If this is landuse change and that change is dictated primarily by economic forces, how good will non-economic approaches be in addressing that cause? The answer to this question strays into the design of policy and the creation of markets. What favours the economic approach is that policy tries to address the different stakeholders in biodiversity decline. Moral approaches could address this issue, but might miss it. Declaring an area to be 'protected' for the sake of biodiversity value may leave some stakeholders disaffected and unwilling to co-operate in the protection plan. Devising economic incentives to enable all stakeholders, as far as possible, to be better off with the protection plan than without defines the economic approach to policy design, but it is also possible to imagine an approach based on conservation because it is 'right' which also designs such a policy package.

IV. ELICITING VALUES: DELIBERATIVE AND INCLUSIONARY PROCEDURES

4.1 Introduction: forms of deliberative procedures

A recent literature has placed emphasis on *deliberative and inclusionary procedures* (DIPs) for the elicitation of individuals' values. DIPs involve the testing of *stakeholder preferences* through the use of consultative procedures over and above those that would normally be associated with decision-making in democratic societies. Some commentators argue that the stimulus to such procedures comes from a dissatisfaction with prevailing institutions as a means of involving stakeholders, obstacles to the articulation of stakeholder preferences, and a concern about imbalances of power to determine outcomes. The types of institution being considered are:

> *Focus groups*: a group of no less than six and no more than twelve persons, theoretically selected at random from a general or a target population. Preferences are elicited through discourse mediated by 'moderators'.
>
> *Citizens' juries*: a selection of individuals who are asked to deliberate on a policy issue, the request usually, but not always, coming from an agency that has the power to act on any outcome. As in a trial, jurors are presented with evidence and can examine 'witnesses'. Questions and who is questioned, and the general procedure are determined by a separate group of stakeholders. As with focus groups, a moderator acts as rapporteur and writes a report for the commissioning agency. Groups again tend to be small, about one dozen people as in a conventional trial jury.
>
> *Consensus conference*: similar in many respects to a citizens' jury, a consensus conference consists of a panel of around a dozen people who are set a specific question, usually aimed at a broad-ranging scientific or technological issue. The conference has no moderator and

is left to establish its own procedures for working based on an initial package of information. Experts may be examined and a final report is written, usually aimed at the general public. Consensus conferences have emerged in Denmark as the primary example, with other countries having experimented with this procedure.

Deliberative polls: a large sample, perhaps up to 500 people, is invited to attend a special location for several days. Fees and expenses may be paid. The whole sample is subdivided into smaller groups who determine the issues to be examined by the group as a whole. Questionnaires are handed out at the beginning and end of the sessions to determine shifts in opinion during the process. A moderator is involved.

One feature of deliberative procedures is that the group context allows interaction between participants who may therefore become informed by listening to others' views. The process is said to be 'transformative' because values at the beginning of the process may differ from those at the end. Obvious risks include the possibility that views may be unduly influenced by dominant personalities or more articulate individuals. Opinion is divided on the scale of these risks. Small groups can readily be dominated but each individual may be less inhibited about speaking. Large groups are probably less at risk of domination but may produce 'silent majorities'. Issues of representation arise. Small groups may set out to be statistically representative but limitation on size means that samples are necessarily non-random. There are also substantial variations in what is regarded as a legitimate participant. Economic approaches to policy appraisal, such as cost-benefit analysis, define stakeholders in terms of gainers and losers, and the relevant unit is the individual as user, taxpayer or holder of a non-use value. DIPs tend not to be precise about what constitutes a stakeholder, but many interpret the term to mean those with an interest in the decision. This may involve NGOs, quasi-government organisations, executing agencies, political bodies etc, and the general public may not be involved, although they often are. Representation is also a problem with most forms of DIPs since they involve significant time inputs which many members of the general public may not be able to offer, especially if the procedure runs across several days. In some cases, participants answer advertisements in newspapers so that representation may be further limited by self-selection in the sample. Problems of self-selection are well established in other forms of public consultation, e.g. public inquiries. Concerns about non-representation have been voiced in several quarters. Deliberative procedures have been used in Local Agenda 21 discussions in several countries.

There is also a widespread debate on what values should be elicited. Those who believe that decision-making procedures should reflect 'transformative' values - values which are articulated in the presence of factual information, debate and information about how others think - will favour deliberative procedures. Those who believe that the relevant values are those akin to those that would be expressed in a market place, with available information and comparatively little procedure on arguments for or against the object of value, will favour stated preference techniques. DIPs and stated preference procedures are not exclusive. Stated preference techniques (see Chapter VII) can also be used as deliberative procedures; a practice that is, however, more common in developing than developed countries.

4.2 Deliberative procedures: advantages and disadvantages

Table 4.1 lists some of the advantages and disadvantages of DIPs by comparing them to stated preference techniques which themselves are designed to elicit attitudes and values. Stated preference techniques, however, also include questions relating to the respondent's willingness to pay for the asset in question, or questions requiring a ranking of options such that willingness to pay can be inferred. Clearly, since stated preference techniques are themselves the subject of debate, the issues in Table 4.1 relate only to value elicitation and the policy relevance of the procedures used. Moreover, there is no reason to suppose that the two procedures are exclusive of each other, many stated preference procedures involve major elements of DIPs.

Table 4.1 Deliberative and Stated Preference Procedures Compared

Type of issue	Deliberative procedure	Stated preference
Aim	To elicit stakeholder preferences, including ‘transformed preferences’.	To elicit preferences of random sample of the population. Sample may be revisited at a later date to check for consistency of preferences across time, although this is not very usual.
Representation	While efforts are made to make it random, samples are often non-random due to (a) small size of group; (b) broader definition of stakeholders; (c) need to have meetings. Occasionally, general public appears to be excluded as stakeholders.	Samples designed to be random with minimum sample size of 200-300 recommended and 1000 considered best. Sample excludes non-public stakeholders. Short time taken by interviews means sample less likely to be self-selecting.
Biases	Risks of domination by articulate or dominant persons due to interactive context.	Various forms of bias with respect to WTP: e.g. starting point, instruments, embedding.
	Risk of responding in a pleasing fashion to commissioning agency’s interests (‘yea saying’).	Risk of ‘yea saying’.
	Risk that preference stated in group context is changed when away from group.	Hypothetical bias: i.e. stated preferences may not be true.

Table 4.1 continued over page

Table 4.1 continued

Common features	May arouse unrealistic expectations about action following such procedures. Focus group may be used in citizens' juries. Discourse. Elicitation of attitudes, general values. Other biases inherent in opinion-seeking.	Unrealistic expectations a possibility but similarity to an opinion poll tends to avoid this. Focus groups are included in all sound stated preference procedures. Included in some larger stated preference studies. Included in preamble to stated preference studies. Other biases inherent in opinion-seeking.
Distinguishing features	No requirement to state preferences in quantitative form In some forms, witnesses may be examined so that expert opinion is used to guide preferences. Suited to use on the world-wide web.	Stated or inferred willingness to pay (accept) measures are obtained. Scenario definition may give some expert information but of a limited kind.
Cost	Estimates difficult to come by but could be \$20-35,000 for citizens' juries and deliberative polls, and \$120,000 for a consensus conference.	Costs range from \$20,000 to \$500,000 for major studies.

It can be seen that DIPS and stated preference techniques have many features in common. Box 4.1 provides an example of DIPS in Switzerland.

Box 4.1 Deliberative procedures illustrated: Swiss referenda

In common with reforms undertaken in other OECD countries, Swiss agriculture has undergone a transition towards market liberalisation that has challenged the established pattern of production and farm support. The Swiss transition can be attributed to a more direct expression of citizen preferences for traditional production patterns expressed through the traditional ballot system for important democratic and policy changes. A study by Gunter *et al.,* (2002) is an example of the 'provider gets' principle wherein referenda are interpreted as willingness to pay decisions for environmentally based support to specific farmers. Referenda are said to avoid the 'principal agent problem' of representative democracy and farmers are made aware of the real public demand for non-market goods and the potential danger in terms of access to support (given the direct ballot system) of contravening the rules of the principle. This system is effectively a form of expressed demand or willingness to pay for environmental goods. The process of direct democratic approval for agricultural support is akin to a valuation decision that reveals the economic value for specific environmental improvements more by implication than by direct expression of willingness to pay. The public votes for a package of specific environmental support and willingness to pay for specific environmental goods is inferred from the breakdown of the money for mandated programmes. Thus it is possible to identify implicit values for Alpine meadows and agricultural intensity of land use, the loss of pastures and meadows, traditional orchards hedgerows - all of which are important nesting and breeding sites for birds and butterfly species. The authors make a comparison of these values with results from recent contingent valuation studies on landscape and species preservation, and find that the implicit values are in fact much higher then the explicit contingent valuation results. The conclusion is that since the policy environment for agriculture is in a period of transition, and since it is difficult to paint a reliable scenario for respondents to value using a contingent market, the referendum process carries as much validity as any individual preference model. This is especially true if expert opinion is divided as to what it is about biodiversity that individuals should be asked to value. In both cases it is important to stress that voters may have had only a limited idea of the question at stake. At best, the question can be framed in terms of a decisions about patterns of agricultural production rather than about specific levels of diversity. The authors suggest that how referenda come to be framed is highly dependent on the political equilibrium. It is unclear to what extent this is a valuation model that can be replicated in countries with alternative democratic traditions. The use of direct democratic participation in decision-making can be seen as central to a growing area of informed decision-making using deliberative procedures, wherein a cross section of stakeholders reach consensus on policy issues.

V. VALUES AND TIME

5.1 Biodiversity as a long-term asset

Biodiversity contains some characteristics that make it different to many other goods. In so far as biodiversity generates instrumental aesthetic benefits and values, it is arguable that future generations might not share those values. Hence biodiversity in the future could be less valuable than it is today. Equally, future generations might care even more about the aesthetics of biodiversity than the current generation. The wants, tastes and preferences of future generations are unknowable in the sense that they would be very difficult, if not impossible, to forecast hundreds of years ahead. Decisions about biodiversity conservation could therefore focus on the instrumental value for current generations only, ignoring what future generations want because their wants are unknowable. There are several reasons why such a view would probably lead to the wrong decisions being made about biodiversity.

The focus that governments in all countries now place on *sustainable development* suggests that policy-making is at least a little more directed towards the longer term than hitherto. Whether future generations have 'rights' or not is debated by philosophers, but it is certainly the case that many people care about the nature and quality of the assets that future generations will inherit. Hence making decisions on the basis of current preferences and concerns is likely to understate the long-term benefits of biodiversity.

A second issue is whether the wants of future generations are truly 'unknowable'. Generations overlap: children and grandchildren exist today and the current generation can speak to them. In this fashion it is unlikely that what the immediate generations to come would like by way of biological assets is truly unknowable.

Third, in so far as there is a concern for the well-being of future generations, that well-being will depend on the assets and technology available to them. One of those assets is biodiversity, so that reducing the size of the asset now would deprive future generations of the *option* to utilise those resources. Essentially, allowing biodiversity to decline reduces the asset base on which future generations' well-being will depend. This is the logic lying behind the notion that it is best to 'keep options open'. Moreover, this approach has added value when the full value of the asset is not known, as is the case with biodiversity. Conservation should result in more and better information about the value of biodiversity. Reducing biodiversity means that this yet-to-occur information will be lost. Avoiding losses is especially important in contexts where decisions are *irreversible*. Reversibility implies that, if a mistake is made, it can be corrected. Irreversibility precludes that corrective process. At least part of the stock of biodiverse assets is irreversible. Habitats can be destroyed and, in principle, recreated, but the chances of recreating assets that have been the product of thousands and millions of years of evolutionary processes are remote.

Even if values are only instrumental, there are nonetheless powerful reasons for conservation for the benefit of future generations. If values are 'objective', in the sense described in Chapter III, then time cannot change those values. Since they are part of the asset they are a 'fact' rather than a value conferred by a valuer. Those who believe in objective values will therefore have an added reason for conservation, namely to conserve the value that resides 'in' biodiversity.

5.2 Time and decision-making

While there are strong reasons to conserve biodiversity for future generations, it remains the case that decisions are made by the current generation. Their concern for the future is tempered by the observable fact of 'time preference', the preference that individuals have for securing benefits now rather than later. Individuals are said to 'discount' the future, and discounting can quickly produce very harmful effects on assets that have long-term benefits.

A generalised formula for discounting is

$$W_t = 1/(1+r)^{\alpha(t)} \qquad (1)$$

where W_t is the weight to be attached to a cost or benefit in year t, and is known as the *discount factor* and r is the discount rate, and $\alpha(t)$ is function of the perception of the speed at which time passes. For conventional ('exponential') discounting, $\alpha t = t$, so $\alpha=1$, so that

$$W_t = 1/(1+r)^t \qquad (2)$$

To see how long run assets can be damaged through the process of discounting, a simple example will suffice. Discount rates in a typical OECD country would probably be in the range of 5-8% in real terms (i.e. net of inflation). For an asset with a value of $1 billion in 50 years' time, the effect of discounting at 8% is to make that value equal to just $21 million today. Put another way, its value in 50 years time is reduced by a factor of just under 50 because of discounting. The higher the discount rate, the bigger this division factor is.

While equation (2) is the discount factor formula that can be found in economics textbooks, there is in fact no particular reason to suppose that discounting should proceed in this way. It is necessary to distinguish money goods from goods other than money. If the good in question is money, the existence of positive rates of interest in the economy does imply exponential discounting. Essentially, if the market rate of interest is i, then $1 next year is not worth the same as $1 now because $1 now can be invested at i% to become $(1+i) next year. Conversely, $1 next year must be worth $1/(1+i) now. If there is a constant market rate of interest, discounting on this argument implies an exponential discount rate. At the level of the economy as a whole, individuals' time preferences help to determine the market interest rate. Individuals adjust their consumption and savings so that, *at the margin*, they discount at the interest rate. Just as prices are measures of marginal valuation, so interest rates are measures of the marginal time preference for money.

When the good in question is not money, then, given that money is nonetheless the standard of value, there is no measure of the discount rate that is independent of the money valuations of the good in the relevant periods. Suppose that X is willing to pay $1 for a unit of clean air today and $0.99 for clean air to be enjoyed tomorrow, the marginal time preference rate (MTPR) for clean air is 1%. If the MTPR in money is 5%, the money value of clean air is increasing at 4% per year. In general, the MTPR for different goods will be different, simply because money valuations for goods change over time. It is a convention that money is used as the unit for MTPR and that everything else is treated as changes in money valuations. But having adopted this convention, we have to use the MTPR for money (which equals the interest rate in a well functioning market) as the discount rate for all costs and benefits once these have been measured in units of money.

But if the focus is on the way in which individuals discount the future there are no a priori restrictions on the nature of discounting. Indeed, it is possible that people may discount the future differently for different goods and that they may even discount the future at a negative rate (e.g. preferring to 'get the worst over now'). A lot of interest is now being shown in *hyperbolic* discount rates. The generalised function can be expanded slightly:

$$W_t = 1/(1+r)^{\alpha(t)} = 1/(1+gt)^{h/g} \qquad (3)$$

where h measures the speed of an individual's time perception. If h = 0, time periods pass infinitely quickly. If h = ∞ time is perceived as not passing at all. g measures the degree of departure from the standard (exponential) discounting model. As tends g to 0, so W_t approaches the conventional discount function. Cross multiplying:

$$(1+r)^{\alpha(t)} = (1+gt)^{h/g} \qquad (4)$$

and hence

$$\alpha(t) = h.\ln(1+gt)/g.\ln(1+r) \qquad (5)$$

From this general formula various *hyperbolic* discount factors can be derived. Hyperbolic discounting produces *lower discount factors* in the near term than the conventional model, and hence *higher discount rates*, but *higher discount factors* than the conventional model, i.e. implicitly *lower discount rates* , for periods of time further into the future. Hence if biodiversity is thought to have long run importance, it is better to manage it using hyperbolic discounting than conventional exponential discounting.

5.3 Discounting and the very long term

Discounting appears not be an issue for those who do not believe in instrumental values. Discounting is simply a way of expressing the preferences that people have for the present over the future. If value does not reside in the valuer but in the object itself, then the rate at which individuals discount the future is irrelevant. In fact, discounting is unavoidable since the neglect of discounting, or treating it as irrelevant, is formally equivalent to using a discount rate of zero per cent. Zero is a number just like any other number. Applying a zero rate implies indifference between the occurrence of benefits now or in the very distant future, an indifference that, on the objective value approach, arises because of the intrinsic nature of value. Zero discounting is problematic. If biodiversity is thought to yield greater absolute benefits to people in 1000 years time, then zero discounting would imply that the current generation should sacrifice everything now in order to generate those benefits in the distant future. The argument could quickly become a justification for impoverishment now for the sake of generations in the very distant future[13].

But even for instrumental value there are arguments that suggest the discount rate for the very long run should be very low. One argument in defence of this position is advanced by Weitzman (1998). As he notes, 'to think about the distant future in terms of standard discounting is to have an uneasy intuitive feeling that something is wrong, somewhere'. It is not just a matter of lowering the value of an event 50 years from now compared to one happening now, but discounting also makes a substantial difference between the way we view an event 400 years from now compared to one 300 years from now. Yet few people would make the distinction between events that far ahead in time. Weitzman observes that at such times in the distant future we cannot really have much idea what the discount rate is. One way of underlining this is to think of a discount rate as a rate of interest or return on capital assets (the so-called 'marginal productivity' approach). Then, 300 years from now, the rate of return on capital could be enormous or, if dire predictions about the world's environmental future come about, could be negative. Hence the discount rate becomes a random variable, a magnitude that could take on any number of values in a random fashion. This suggests taking some sort of average, but Weitzman observes that what should be averaged is not the discount *rate*, r, but the discount *factor* $1/(1+r)^t$. The discount factor is the weight being applied to gains and losses in different periods.

The difference between choosing discount rates and discount factors to average is formidable. Suppose the range of probable discount rates 100 years from now is 1-10% each with an equal probability of occurring. Then the average would be 5.5%. But the discount factors would range from 0.00007 to 0.37. If these occur with an equal probability then their average is 0.05924. The discount rate that corresponds to this discount factor 100 years from now is given by $1/(1+r)^t = 0.05924$, or r = 2.8%, half the rate secured by average discount rates.

13 More generally, the example illustrates the confusion involved in trying to address problems of intergenerational equity through the adjustment of a 'price', i.e. the discount rate. Discounting is about efficiency, not equity, and adjusting efficiency prices to deal with equity concerns is regarded by many economists as a misconceived policy. This does not mean that intergenerational equity should not be addressed, only that there are better ways of addressing it.

Weitzman's proof that the discount factor should be treated as the random variable is complex but the intuition is attractive. The discount rate that is implicit in the process of averaging discount factors over time steadily decreases with time (i.e. as t becomes large). Weitzman suggests that, in practical terms, near-term discount rates should be 3-4%, but 2% for periods 25-75 years from now, 1% for periods 75-300 years from now and zero % for 300 years plus. This is, in effect, a form of hyperbolic discounting.

VI. ECONOMIC VALUES: THE BASICS

6.1 The nature of economic value

In terms of the classifications introduced previously, economic value is an instance of an instrumental value. Economic value is linked to cost-benefit analysis (CBA), although its uses range far more widely than this. CBA is a procedure whereby for any change in the *status quo* the benefits of that change may be compared to the costs. A benefit is defined as any positive change, i.e. any gain, in human well-being (also known as welfare or utility) regardless of who secures that gain. A cost is defined as any loss of well-being regardless of who suffers that loss. Note that gains and losses are <u>not</u> defined in terms of financial flows. The notion of well-being is far wider than any change in cash flows.

The basic rules of CBA are as follows:

a. for a situation in which there is only one policy option relative to the status quo - accept a policy (project, programme) if benefits exceed costs.

b. for a situation where policy options are mutually exclusive and one must be chosen – select that policy with the highest net benefits.

c. for a situation where various policies can be chosen as part of an overall programme of change – accept all policies with a ratio of benefits to costs greater than unity until the available budget is exhausted.

d. To complement item (c), use also rate-of-return on investment where feasible to rank projects.

Within any democratic society two forms of decision-rule can be used to deal with situations where people cannot agree unanimously on a course of action. The first is majority voting, so that if more than 50% of voters approve a course of action that is often sufficient for that action to be undertaken. But majority voting has shortcomings. One of these is that the majority may vote in favour without having particularly intense feelings about the action, whilst the minority may feel passionately against the action. A decision rule that compares 'intensities of preference' may therefore be used to amend majority voting, so that an action that imposes heavy costs on a few individuals may still be rejected even though a majority of individuals would benefit moderately from the action. It will usually be important to have some idea of both forms of preference revelation.

Individuals express preferences about changes in the state of the world virtually every moment of the day. The medium through which they do this is the market place. A vote for something is revealed by the decision to purchase a good or service. A vote against, or an expression of indifference, is revealed by the absence of a decision to purchase. Thus the market place provides a very powerful indicator of preferences. Moreover, market-place preferences have two features of direct relevance to the process of economic valuation.

First, the stronger the preference is for something the greater is the willingness to pay. Thus purchase decisions reflect willingness to pay (WTP) and WTP reflects intensity of preference. This is an important modification of the majority voting system in which all preferences count equally, however strongly or weakly they are held.

Second, WTP is constrained by income (or wealth). When individuals are invited to vote for or against a given action, they may be told that the available alternative courses of action compete against each other for limited funds. But they may also not be told this. Yet if they vote in favour of one course of action, the funds allocated to that action necessarily preclude other actions being undertaken. This is the notion of *opportunity cost*. A voting system that ignores, or risks ignoring, opportunity cost fails to represent available choices in a true fashion. WTP, on the other hand, is automatically constrained by income and so should force individuals to relate their WTP to what can be afforded. There is an underlying value judgement in CBA, namely, that individuals' preferences should count, sometimes known as 'consumer sovereignty'.

6.2 Benefits and consumer's surplus

In economics, WTP has a formal relationship to the notion of demand. Figure 6.1 shows the usual depiction of a demand curve for an individual. The horizontal axis measures the total number of units that can be bought and the vertical axis measures the price. Points on the individual's demand curve show, for each quantity purchased, how much that individual is willing to pay for that last (or *marginal*) unit. For example, the individual is WTP $10 for the first unit, $8 for the second unit, $6 for the third unit and so on. The *total* WTP for three units is $10+8+6 = $24. Hence marginal WTP is given by points on the demand curve and total WTP is given by the area under the demand curve. Suppose the market price settles at $6, then total expenditure is 3x$6 = $18 and this is less than total WTP of $24. The difference between total WTP and actual expenditure, i.e. $6, is the *consumer's surplus*. This is given by the shaded area. Consumer's surplus is therefore a measure of the net benefit to the consumer of buying 3 units at the market price since he/she pays out $18 but 'gets back' $24 in the form of well-being as measured by WTP. The $24 in this case is a measure of the *gross* change in well-being from buying 3 units, and the $6, the consumer surplus, is a measure of the net change in well-being.

Figure 6.1 Demand and WTP

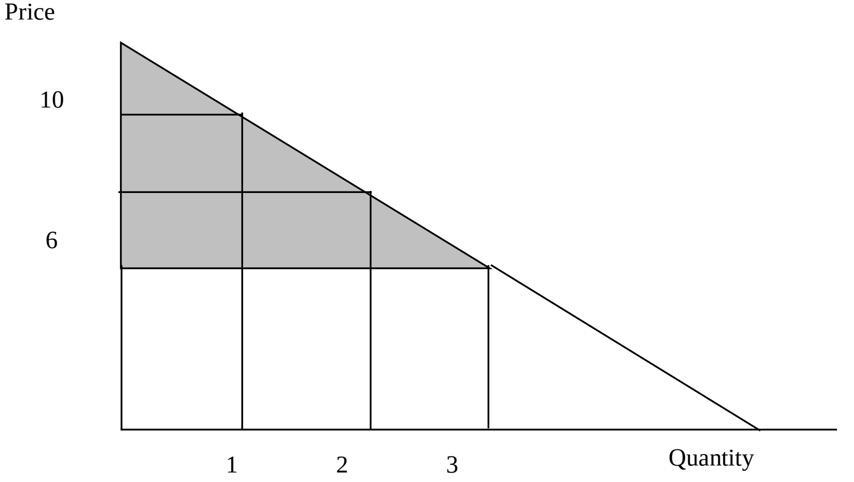


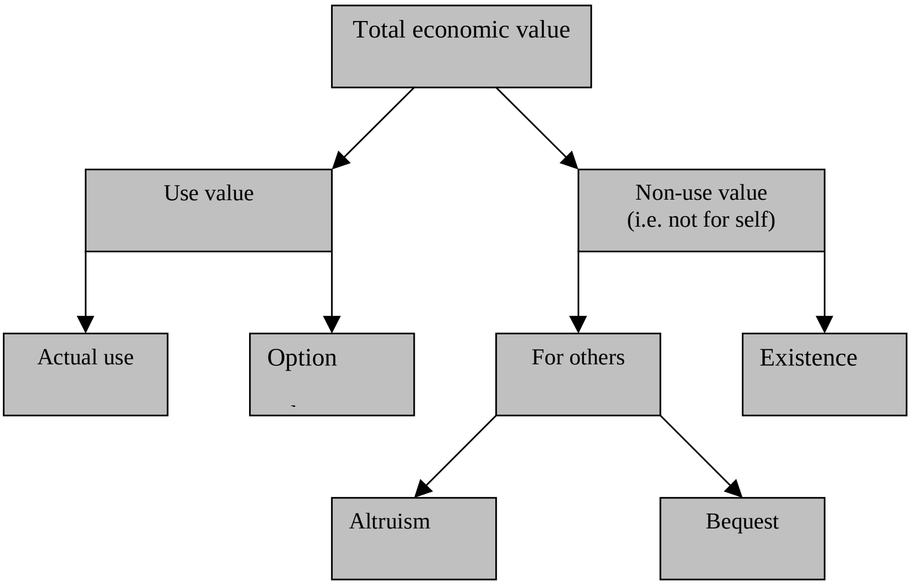
Figure 6.1 shows the situation for a single consumer. Summed across all consumers, the relevant notion becomes *consumers'* surplus as opposed to *consumer's* surplus.

6.3 Total economic value

The net sum of all the relevant WTPs defines the *total economic value* (TEV) of any change in well-being due to a policy or project. TEV can be characterised differently according to the type of economic value arising. Typically, TEV is divided into *use* and *non-use* values. Use values relate to actual use of the good in question (e.g. a visit to a national park), planned use (a visit planned in the future) or possible use. Actual and planned uses are fairly obvious concepts, but possible use could also be important since people may be willing to pay to maintain a good in existence in order to preserve the *option* of using it in the future. Option valuethus becomes a form of use value. Non-use value refers to willingness to pay to maintain some good in existence even though there is no actual, planned or possible use. The types of non-use value could be various, but a convenient classification is in terms of (a) existence value, (b) altruistic value, and (c) bequest value. Existence value refers to the WTP to keep a good in existence in a context where the individual expressing the value has no actual or planned use for his/herself *or for anyone else*. Motivations here could vary and might include having a feeling of concern for the asset itself (a threatened species, for example) or a 'stewardship' motive whereby the valuer feels some responsibility for the asset. Altruistic value might arise when the individual is concerned that the good in question should be available to others in the current generation. A bequest value is similar but the concern is that the next and future generations should have the option to make use of the good.

Figure 6.2 shows the characterisation of TEV by types of value. Differentiating use and non-use values is important because the latter can be large relative to the former, especially when the good in question has few substitutes and is widely valued. In addition, non-use value remains controversial, so that it is important to separate it out for presentational and strategic reasons.

Figure 6.2 Total economic value



6.4 The cost-benefit formula

Taking the simplest case only, i.e. where there is an accept/reject decision for a project or policy, the final formula can be stated as:

$$\Sigma_t \Sigma_i a_i. (B_{i,t} - C_{i,t})(1+r-p)^{-t} > 0 \quad (1)$$

The formula states that benefits (B) and costs (C) need to be summed across all relevant individuals and across time. The notation ‘a’ refers to the weight that is to be attached to each $1 accruing to different individuals. In the simplest case, a = 1. But some individuals may be poor or vulnerable and it may be thought that $1 accruing to them is socially more important than $1 accruing to richer or less vulnerable groups. If so, ‘a’ can be varied by socio-economic group, thus introducing an ‘equity adjustment’ to the formula. In equation (1) ‘r’ is the discount rate, assumed to be constant over time (but see Chapter V). Benefits may well rise faster over time for biodiversity than for other goods (there is a ‘relative price effect’) and hence this can be introduced into the formula either by escalating the value B at p% per annum, or, as shown in equation (1) by deducting p% from the discount rate to produce a net discount rate.

Cost-benefit analysis and the monetary valuation of costs and benefits is controversial and the controversies cannot be reviewed here. Some of the issues have already been raised, e.g. the extent to which the relevant value for biodiversity is a ‘moral’ value and, if so, whether moral value resides ‘in’ the biodiversity or whether it is conferred on the biodiversity by the valuer. The notion of intrinsic value is relevant to the notion of existence value since one motive for existence value may be the concern to elicit the intrinsic value of the biodiversity.

This brief introduction to the nature of economic value sets the scene for the subsequent chapters dealing with methods for eliciting economic value.

6.5 Valuing biodiversity as a support function

It is well known that, without biological diversity, human life would cease. Hence biodiversity has a ‘life support function’. But it is not meaningful to ask questions such as ‘what is the value of this life support function?’ As Dasgupta (2000) remarks:

> ‘The value of an incremental change to the natural environment is meaningful because it presumes humanity will survive the change to experience it. The reason (that) estimates of the total value (of the environment) should cause us to balk is that if environmental services were to cease, life would not exist.’ (p106).

Nonetheless, it is helpful to review just what biodiverse systems do by way of life support and economic functions. The Austrian Federal Ministry of Agriculture, Forestry, Environment and Water management (2000) has reviewed the biodiversity of the Austrian Alpine Region from this perspective. A distinction can be made between what a set of biodiverse regions already do, and what, potentially, they could do. In terms of current uses, the Alps already support a substantial tourism industry, provide protective ecological services (e.g. water and soil protection) for the region and for regions outside the Alps, provide the basis for culturally and socially valuable rural communities, and protection against natural hazards. The study suggests that the monetary valuation of all these functions cannot be estimated with any accuracy, with values ranging from a few billion Austrian schillings to over 1000 billion schillings. Monetary valuation is better reserved for specific features of the Alpine Regions, e.g. national parks. This is in keeping with the Dasgupta quotation above which distinguishes between the value of changes and parts of an entity and the value of the whole entity. In terms of potential value, the study makes two interesting observations. First, the Alpine region is rich in fresh clean water, an increasingly valued commodity in a water-scarce world. In the future, perhaps the provision of saleable water could rival the importance of the tourist industry. Second, a substantial economic activity is based on treating the region as a free good, e.g. television, films, photography. These users make no payment for the valuable biodiversity they utilise, and it is suggested that there might be a ‘copyright in nature’ whereby such users would have to pay a fee each time such uses are made.

VII. ECONOMIC VALUATION METHODS BASED ON MARKET PRICES

7.1 Introduction

Economic approaches to biodiversity valuation consist of three procedures:

a. Using market prices where the prices occur in the market for the biodiverse asset and where prices are 'revealed' in some other market- the *revealed preference* approach.

b. Using willingness to pay estimates derived from questionnaires - the *stated preference* approach.

c. Using values 'borrowed' from existing studies - *benefits transfer.*

This section considers market based and revealed valuation methods.

7.2 Market prices

Market values for biological resources are perhaps the most obvious argument for conserving habitats - and hence biodiversity - threatened by some alternative use. The availability of such markets will however depend on the costs of harvesting and transportation and their proximity of population centres (Pearce and Moran, 1994). Hence, biodiversity that is geographically remote may have a low market value in terms of its direct use. Equally, however, the more remote the biodiversity the less likely it is to be under threat and the less value there is for the alternative use of the land occupied by the biological resources.

There are three valuation approaches based on market values:

– The observed market value and related goods approach.

– The productivity approach.

– Cost-based methods including replacement cost.

These methods rely on the availability of market price and quantity information to derive total values. The productivity, or production function approach requires more analysis to establish a physical relationship between some environmental change, or 'dose' (e.g. deforestation), and an impact or response that can be associated with a monetary value (e.g. downstream flooding or the health of an estuarine fishery). The nature of the relationship between the dose and a response is sometimes complex. Generally, this form of integrated assessment will require ecological expertise. The productivity approach is also the framework that is implicit in discussion about ecosystem services. The degradation of systems will lead to the loss of functions that are ultimately economically relevant if not immediately amenable to valuation using market prices or stated preferences. The replacement approach values the asset according to the cost of replacing it. This is not strictly a valid procedure since the issue is whether replacement is worthwhile or not. The cost of replacement therefore needs to be compared to the economic value of the replacement. Since these methods are based on the ruling market price, they generally do not provide a measure of consumer surplus (see Figure 5.1 in Chapter V) or non-use benefits.

7.3 Observed market and related good prices

Hundred of studies demonstrate the values of naturally occurring products (for reviews see Pearce, Moran and Krug, 1999; ten Kate and Laird, 1999). Examples include genetic material for agricultural products and drugs, minor forest products, etc. It is important to note that market prices used should be adjusted where necessary to reflect economic values. Necessary adjustments can include:

a. The difference between gross and net value, i.e. deducting production and transport costs from the observed market price to arrive at the net value of a product.

b. Correcting the market prices for any known price distortions or policy failures (e.g. taxes and subsidies) that affect the output itself and any inputs (e.g. labour) that produce the output. If products are traded internationally it may also be necessary to convert ruling domestic prices to border equivalents (i.e. world prices). Correcting for externalities -i.e. the harmful effects of use - may also be necessary. In particular, prices of harvested products should bear some relationship to sustainable yields.

Box 7.1 provides an example in Mexico applied to the “Mamey”.

Box 7.1 Enriching the rainforest with native fruit trees: a case study of Mexico

Natural forest stands in tropical and sub tropical regions are highly diverse ecosystems. But their conservation is often at a high opportunity cost in terms of agricultural development. Consequently there is a focus on measuring the financial and economic returns to forest conservation. Reduced impact logging methods and forest enrichment – the use of productive fruit species intercropped with existing species – are suggested management options that might enhance the economic returns to forest conservation. Both methods depend on reliable forecasts of the values of both timber and non-timber forest products. A study by Ricker *et al.*, (1999), presents an ecological-economic approach to measuring the likely return to an indigenous tree fruit species *Pouteria sapota* (‘mamey’) interplanted into the natural forest in Mexico. The ultimate economic objective is to calculate the Net Present Value (NPV) of fruit production per hectare for comparison with the likely returns to alternative agricultural activities. The NPV is given by:

$$NPV = \sum_{i=1}^{MA} [F_i S_i (P - C) e^{-rt}] - K$$

where i is the age of the timber (harvest is once a year), and planting occurs at i =0. MA is the maximum age of production, F is the expected annual fruit yield of the tree at age i, S indicates the survival probability of trees in a hectare sample, P and C are unit price and costs for tree harvesting, K is a separate cost element for planting and establishment, and r is a discount rate.

The basic calculation requires information on the average annual fruit yield, survivorship of planted fruit trees, market prices for the fruit, and the appropriate costs of harvesting and transporting them to markets and costs of initial tree establishment. Because it is difficult to determine the age of trees in lowland tropics, the study develops models of tree growth in order to set a time horizon on the total amount of fruit produced by a tree over its lifetime. The tree growth to maturity profile in turn sets a profile for the production of fruit (per hectare), subject to an estimated rate of mortality for the sample of trees in that hectare.

Box 7.1 continued over page

Box 7.1 continued

In total the authors identify eleven parameters that are central to the combined ecological and economic model that predicts the present value of fruit production. Besides the choice of an appropriate discount rate, the essential valuation issue is one of determining accurate values for these parameters rather then valuing the biodiversity that is assumed to be intrinsic in forest conservation. The most prominent valuation method used is the correct interpretation of market prices for fruit. But the authors suggest that NPV is in fact most sensitive to the biological parameters linking fruit yield to tree age. Nevertheless prices and costs are important and the authors provide an average local market price for fruits over a two-year period. They also provide a detailed breakdown of harvesting and marketing costs. They are less specific on some elements that might characterise a more rigorous cost-benefit analysis – specifically, the assumptions that market values of labour inputs are their true scarcity values and the market price implications for the widespread adoption of mamey cultivation. These shortcomings are common to other well-documented non-timber forest studies. This signals a note of caution about the use of market prices for valuing non-market goods. Such prices can vary. They will also be based on some assumption that harvesting is maintained at some rate that is within the limits of sustainable production.

7.4 The productivity approach

The productivity method values biological resources as inputs by observing the physical changes in environmental quality and estimating what differences these changes will make to the value of goods and services that are marketed (e.g. agricultural and forest products and fish). An example is a change in wetland size that leads to a change in water quality that reduces the quantity of fish caught. This lost market value can be estimated using market price information. The difference in the value of output resulting from the change being the value attributed to the amount of lost wetland. The production function is the formal representation of this relationship between the change in environment as an input and the change in the production of a marketed output. The method can be undertaken with varying degrees of rigour applied to the derivation of the physical relationship (or dose-response) between the inputs (environmental assets and other man made inputs like capital and labour), and the valued output.

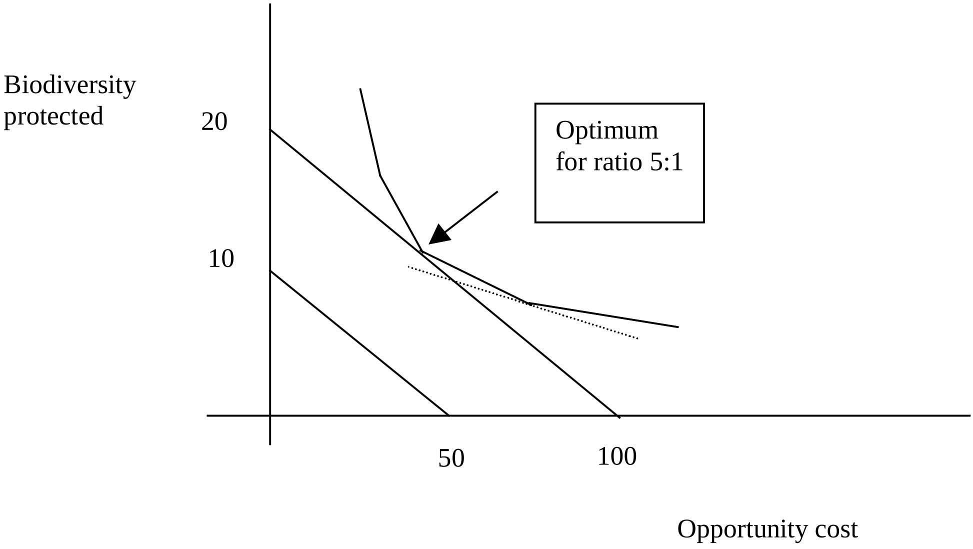
7.5 Cost-based methods

Cost-based valuation approaches include replacement costs, restoration costs, relocation costs and preventative expenditure approaches. Essentially these methods assess the costs of different measures that would ensure the maintenance of the services provided by the environmental asset that is being valued. This methodology is often used ex post when considering the replacement of a lost ecological function such as storm flood protection or soil fertility. Cost-estimates are then used as a measure of the non-market benefit in question. Strictly, measuring benefits by looking at costs is not theoretically correct, as costs need bear no relationship to WTP or demand. Costs as a measure of benefits will always produce a benefit cost ratio of unity, so the method does not give guidance on the efficiency of investing in the replacement. The replacement cost method has recently received much attention in papers that have attempted to estimate the values of ecosystem services (Costanza *et al.*, 1997; Pimentel *et al.*, 1997 and Ehrlich and Ehrlich, 1996). As Bockstael *et al.*, (2000) point out, the replacement cost is only a valid measure if three conditions are met: 1) that a human-engineered system provides functions that are equivalent in quality and magnitude to the natural function; 2) the human-engineered solution is the least cost alternative way of performing the function; 3) that individuals in aggregate would in fact be willing to incur these costs if the natural function were no longer available. Since these conditions are rarely achieved, the simple use of replacement costs is rarely accurate (see Pearce [1998] for a critique of the Costanza *et al*. estimates). Nonetheless, such approaches are widely used. The replacement or restoration cost is, for example, implicit in the 'public trust' doctrine in the USA as it relates to certain natural resource damage costs. Box 7.2 provides an example in the Czech Republic.

Opportunity cost-analysis is also fairly prominent in market price approaches to economic valuation. Two contexts for opportunity cost-analysis can be distinguished. In the first, some indicator of biodiversity is traded off against the (opportunity) cost of biodiversity conservation. In the second, the biodiversity indicator and the cost-indicator are supplemented by other criteria that may be thought relevant to the conservation decision (*multi-criteria analysis*).

An example of the former approach is given by Faith (1997). The aim is to compare the contribution to biodiversity that a given additional area of protection will provide, to the value of the forestry resources that would be displaced by protection. The additional biodiversity achieved has to allow for complementarities between protected areas, e.g. what is achieved by protecting area N depends on what is adds to the biodiversity in areas A…M. Iterative procedures are needed to establish what each alternative area of protection would add to overall diversity. Costs are weighted relative to biodiversity and this results in a series of 'equal net benefit contours' - see Figure 7.1.

Figure 7.1 Trade-off curve



In Figure 7.1 a particular weighting is shown such that costs are weighted relative to biodiversity by a factor of 5. By plotting the various cost/protection outcomes for a given budget, points in the diagram space can be entered. The tangency of the equal net-benefit contours with the resulting curve produces the optimal solution, where optimum here means minimum cost for a given level of protection. The optimal point will change if the ratio of values between opportunity cost and protection are changed. The dashed line, for example, shows what would happen if the ratio was about 16:1. The procedure has been used to analyse trade-offs in forest areas in New South Wales. The main focus of interest is in the resulting curve rather than the weighting procedures, i.e. the main interest is in tracing the piecewise linear curve in Figure 7.1. Even without weightings it is possible then to find combinations of protected areas that are superior to those found simply by focusing on biodiversity value alone, where superior means that opportunity costs are minimised. To choose between points on the piecewise linear curve, however, does require weightings. Sensitivity tests were conducted to see how the choices varied with different weightings.

Assigning high weights to forest costs enables the identification of areas that would be protected even under this scenario. Similarly, identifying areas that would not be afforded protection even if forestry had low value provides another benchmark for prioritisation.

Box 7.2 Valuing ecological systems in the Czech Republic using restoration costs

A study in the Czech Republic (Sejak et. al.2002) has attempted to value the ecological assets of the whole country using costs of restoration as the valuation procedure. As the main text discusses, restoration costs are not a value of damage done. But they have a role to play in contexts where the pre-damage situation is regarded as high priority goal to be achieved (a 'strong sustainability' notion) or where some form of public trust doctrine permeates the legal system. Ecological biotypes are determined and each biotype characteristic is given a 'score' on the scale of 1 to 6. Thus matureness would be one characteristic, as would naturalness. Each of these characteristics can be given a score on the basis of expert (ecological) judgement. The scores are then aggregated and the aggregate score is multiplied by a cost-per-unit-score. Thus the total restoration costs for each biotype can be estimated. A GIS system was used to map the economic values attributable to the whole of the Czech Republic territory. The resulting aggregate value of CZK 27 billion is a factor of twenty times the Czech Republic GNP, the former being a value of the natural capital stock and the latter a measure of economic flow. More information, including maps, can be found at http://www.ceu.cz/gis/cmapa_en.

7.6 Revealed preferences

Revealed preference methods use observed behaviour to infer the environmental value. Contrary to the market price approach, however, the relevant prices are in markets that are affected by the non-market asset. The approaches include traditional *travel cost* models of *recreational use*, *random utility* models, *hedonic* models, and *averting behaviour* models. Each of these methods relies on a surrogate market that provides a 'behavioural trail' to identify the environmental value of interest. Because these values are revealed in real (rather than hypothetical) behaviour many economists and decision-makers are more comfortable with their predictions. The disadvantage of these approaches is that they are limited in terms of the biological resources to which they apply. The methods are also demanding of data.

7.7 Revealed preference: travel cost methods (TCM)

Many natural resources, such as lakes and forests, are used extensively for the purpose of recreation, which includes wildlife and landscape appreciation. But it is often difficult to value these resources when no prices exist with which to estimate demand functions (i.e. willingness to pay functions). However, travel cost models take advantage of the fact that in most cases a trip to a recreation site requires an individual to incur costs in terms of travel and time. Different individuals must incur different costs to visit different sites. These implicit prices can be used in place of conventional market prices as the basis for estimating the value of recreation sites and changes in their quality. The visitors' travel costs act as a proxy for the 'price' they are willing to pay. Clearly, because travel cost models are concerned with active participation, they only measure the use value associated with any recreation site - non-use values must be estimated via some other technique, such as stated preference techniques.

Two perspectives on travel cost are possible. Simple travel cost models attempt to estimate the number of trips visited to a site or sites over some period of time, perhaps a season. Random utility models consider the specific decision of whether to visit a recreation site, and if so, which one. Two variants of the simple travel cost model are considered here [for a more detailed discussion see Freeman (1994) or Herriges and Kling (1999)].

The simple travel cost-visitation model can be used to estimate (representative) individuals' recreation demand functions. By looking at how different individuals respond to differences in monetary travel cost it is possible to infer how they respond to changes in price. The usual assumption economists make is that less of a good is demanded as its price increases. By extension, the number of visits to a site would normally be inversely related to the size of travel cost. The information on peoples' responses to their travel costs is used to draw up a demand curve for the site. The individual's valuation of the recreation site is the area under his or her demand curve, so that the total recreation (use) value of a site is simply the area under each demand curve summed over all individuals. As well as providing estimates of the value of the site itself, the approach can provide values for environmental quality attributes of a site. This is possible using observations of how visitation rates to a site change as the environmental quality of the site changes.

The main steps involved in a simple travel cost study are:

1. Choice of the dependent variable.
2. Dividing the area around the recreation site into zones.

3. Sampling visitors to the site.
4. Obtaining visitation rates for each zone.
5. Identifying multipurpose trips.
6. Estimating travel costs.
7. Obtaining a statistical regression.
8. Constructing a demand curve.

Some of these methodological choices can lead to significant changes in estimated consumer surplus. This can make the method somewhat problematic. Taking each step in turn:

1. For the choice of dependent variable there are two options a) visits from set zones around the resource; b) visits made by a given individual. The second option defines the individual travel cost model and relies on collecting trips per annum information from an individual respondent. The first option is concerned with discovering trips per capita. While there is no consensus in the literature on which dependent variable to use, the consumer surplus estimates derived by these different methods have been shown to diverge substantially. In particular, a study by Willis and Garrod (1991) discovered that the individual approach reduced the consumer surplus estimate for the U.K. Forestry Commission estate from £53 million to £8.7 million. Much of the divergence in these results is due to errors made by respondents to individual visit frequency surveys in recalling their past travel.

2. Zoning - the area around the site is first divided into zones, such that the travel cost to the site from each point in the same zone is roughly the same. In the most straightforward cases the zones could be drawn using concentric circles around the site. But the zones could also be irregular contours, or even non-concentric depending on how travel costs varied within the catchment area of the site.

3. Sampling visitors - questionnaire surveys are undertaken amongst visitors to the site. Data are collected on the characteristics of the visitor, the motives for the visit, travel costs, and attributes of the site. Specifically, information is collected on: number of visitors; place of origin; frequency of visit; socio-economic attributes; duration of journey; time spent at site; direct travel expenses; respondents' valuation of time; total population in each zone; other motives for trip; other sites visited during the journey; and environmental quality attributes of the site.

4. Visitation rates - for each zone the number of annual visits (or visitor-days in the case of overnight stays) per head of the total zonal population is estimated from the survey information.

5. Multipurpose trips can be a problem for TCM as it is assumed that people only derive enjoyment from the site they are visiting and not from others they visit along the way.

6. Travel cost-estimation. The main items to be estimated are: direct expenses incurred by visitors in getting to and from the site, including fares, fuel and other incidentals; the value of time spent on the journey, including time spent at the site (see below); entry fees, guide fees and other incidental expenses at the site.

For round trips involving several sites, travel costs may need to be allocated between each site in a pro rata fashion.

7. Statistical regression - multiple regression analysis is used to test the relationship that travel cost (independent variable) 'predicts' or 'explains' visitation rates (dependent variable) i.e. visitation rates are regressed on travel costs and other socio-economic variables such as income, education, etc, to give a 'visitation rate equation'. One typical functional form for such a regression is:

$$V_z/N_z = f(C,X),$$

where V_z is the total number of trips by individuals of zone z per unit of time; N_z is population of zone z; C is travel cost from zone z; X are socio-economic explanatory variables including income. More precisely the function to estimate is:

$$V_z/N_z = a - b.C_z + c.X_z$$

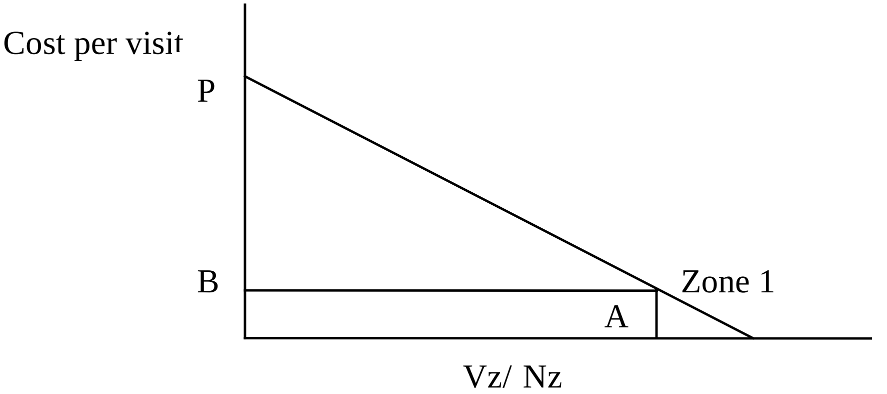
where V_z is visits from zone z to the site, N_z is the population of zone z, C_z is total travel cost from zone z to the site, X is average income in zone z, and a, b, c, are the coefficients to be estimated. Coefficient b is of particular interest, denoting the change in visitation rate as a function of travel cost.

8. Constructing a demand curve. In order to produce a demand curve for the site the estimated visitation rate equation above is used. The assumption is usually made that any increase in travel cost has the same effect on visitation rates as an equivalent increase in a hypothetical admission fee. Points on the demand curve are then found by using the estimated visitation rate equation to compute the visitation rate for a given increase in admission fee (or rather its surrogate, travel cost). This is repeated for successive increases in admission price such that the full demand curve is found. The benefits (consumer surplus) of the site are then found from the area under the demand curve for each zone.

A simplified example is set out below. The visitor rate V_z/N_z is calculated as visits per 1000 population in zone z. For simplicity in each zone the household consumer surplus (cs) for all visits to the site is calculated by integrating the equation of the type: V_z/N_z = a+bC between the price (cost) of visits actually made from each zone (B in Figure 7.2 below) and the price at which the visitor rate would fall to zero. Annual total consumer surplus for the whole recreation experience can be estimated in each zone by first dividing total household consumer surplus (BAP in zone 1) by the zonal average number of visits made by each household to obtain the zonal average consumer surplus per household visit. Multiplying the results by the zonal average number of visits per annum gives the annual zone consumer surplus. Finally, aggregating zonal consumer surplus across all zones gives the estimate of total consumer surplus per annum for the whole recreational experience of visiting the site.

Figure 7.2 Estimating benefits using the travel cost approach

$$\text{C.S.} = \int_{C_z=B}^{P} (\mathrm{a} + \mathrm{bC_z})\,\mathrm{dC_z}$$



Where C_z is cost in zone Z.

The table below provides a guide to these calculations.

Assume the following situation:

Zones	Zonal population (Nh)	Household visits (Vhj)	Average number visits household	Average travel cost per household (Ch)	Consumer surplus per household all visits p.a. ($)	Consumer surplus per household per visit ($)	Total consumer surplus p.a. ($)
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
1	10 000	12 500	1.25	0.16	2.60	2.08	26 040
2	30 000	30 000	1.00	1.00	1.67	1.67	50 100
3	10 000	7 500	0.75	1.83	0.94	1.25	9 400
4	5 000	2 500	0.50	2.66	0.42	0.84	2 100
5	10 000	2 500	0.25	3.50	0.10	0.40	1 000
Total consumer surplus of the whole experience							88 000

Column 1 identifies zones of increasing travel cost. Total population (number of households) in each zone is identified in Column 2. Column 3 shows households visits per zone and per annum calculated by allocating sampled household visits to their relevant zone or origin. The household visit rate shown in Column 4 is calculated by dividing column 3 by column 2. Column 5 shows zonal average cost of a visit calculated with reference to the distance from the trip origin to the site. Demand and consumer surplus estimates using the hypothetical linear demand function (below) are shown in Column 6.

$V_z/N_z = 1.3 - 0.3C_z$.

Consumer surplus for each zone is obtained by integrating the demand curve between the actual cost of visits and that price at which the visitor rate would fall to 0.

$$\text{C.S.(Zone3)} = \int_{Cz=1.83=B}^{P} (1.3 - 0.3C_z)\,dC_z.$$

In column 7 consumer surplus is divided by the zonal average number of visits made by each household to obtain the zonal aggregate consumer surplus per household visit. In column 8 the zonal average consumer surplus per household visit is multiplied by the zonal average number of visits per annum to obtain annual zonal consumer surplus. Finally, annual consumer surpluses are cumulated across all zones to obtain the total consumer surplus per annum for the whole recreational experience.

There are a number of issues which can lead to complications for the simple travel cost model. These problems have been addressed extensively in the literature (see Herriges and Kling, 1999) and the most important are:

1. Time Costs – determining the value to be attached to travel time.
2. Dealing with multiple site visits.
3. Truncation or sample selection bias in dealing with site visitors, and neglecting non visitors.
4. Choice of functional form to represent the relationship in the figure above.

7.8 Application of the travel cost method for biodiversity

The travel cost approach is an important method of evaluating the demand for recreational facilities. The techniques used have improved considerably since the earliest studies were carried out both from an empirical and theoretical point of view. There are however reservations as to its use, particularly concerning the large amounts of data required, which is expensive to collect and process. Furthermore difficulties remain with the estimation and data analysis techniques and so the method is likely to work best when applied to the valuation of a single site, its characteristics and those of other sites remaining constant. The method has limited use for valuing anything other than parks and charismatic species that can provoke travel behaviour. Thus the most credible applications to date have involved national parks, recreational sites and international travel behaviour to visit wildlife parks and reserves (Tobias and Mendelsohn, 1991; Maille and Mendelsohn, 1993; Hanley and Ruffell, 1993).

7.9 Hedonic pricing

The hedonic price method (HPM) uses a different surrogate market to determine values of a non-marketed good. The HPM is based on the idea that a private good can be viewed as a bundle of characteristics, each with its own implicit price, some of which may be non-market in nature. Individuals express their preferences for a particular non-market attribute by their selection of a particular bundle of characteristics. These preferences will be reflected in the differential prices paid for the private good – typically property - in the market. The approach then applies econometric techniques to data on private good characteristics and prices to derive the relationship between the attributes of the good and its market price and from there estimate implicit prices for non-market characteristics. The example most frequently used is that of the housing market. For example, the location of residential property can affect the (non-market) environmental attributes of the property, and potentially, the stream of benefits associated with residence. Neighbourhood features such as air quality, proximity to woodland or water, noise etc. tend to affect the price of the property.

The hedonic approach comprises two main stages (see Garrod and Willis, 1999). First, an equation is estimated to explain house prices or rents as a function of a number of housing and neighbourhood characteristics (including any environmental and cultural attributes of interest). This gives a hedonic price function from which the implicit price of the environmental or cultural attribute can be estimated for each level of the attribute. Second, using the implicit prices faced by each household, an equation relating implicit environmental or cultural prices to the respective attribute levels and various social and economic characteristics is estimated. This corresponds to a marginal WTP function.

The hedonic approach is founded upon a sound economic theory base and is capable of producing valid estimates of economic benefits. However, it has a number of limitations. It relies on the assumption of a freely functioning and efficient property market where individuals have perfect information and mobility so that they can buy the exact property and associated characteristics that they desire and so reveal their demand for the implicit attributes. The approach only reflects impacts to the extent that individuals are aware of them. In addition, a number of statistical problems may hinder its feasibility.

How useful is hedonic valuation when applied to biological resources? There are relatively few rigorous hedonic price studies in the literature and even fewer addressing the value of biological resources. Studies relating to the value of forestry, shoreline and landscape have relied on these attributes being significant in local property markets. In the case of forestry, this amounted to the difference between very distinct woodland types (Garrod and Willis, 1992). In short, the level of biodiversity attributes that can be measured using HPM is limited to facets that show up in the complementary market price, the WTP for housing in most cases. Only a limited subset of biological resources fall into this category and even those that do cannot always be valued if accurate data to describe them is unavailable. It is also important to note that hedonic valuation is essentially ex post and does not capture non-use value.

A closely related application can be found in the area of plant breeding and crop improvement (Evenson, 1990; Gollin and Evenson, 1998). In this context the 'external' value of interest is that of a particular naturally occurring germplasm or genetic trait, which is an attribute of an original crop landrace prior to crop improvement. Such crops existed somewhere before being discovered as an input to world agriculture. The original raw material or trait is ultimately one attribute of a final product that combined a number of other production factors: the returns to which must be stripped out to reveal the base resource value. This value is of interest for two reasons. First as the basis of a return to indigenous cultivation, perhaps as part of a benefit sharing agreement. Second, to determine the value-added, by steps on the path of informal and formal breeding inputs that has led the crop to its current status. Determining value-added by steps provides particularly important information for the CGIAR[14] group when information on the returns to publicly funded research is at issue. However, like its analogue in the housing sector, the analysis required to untangle such values is extremely data-demanding and generally limited to crops that are globally well-documented. Gollin and Evenson (1998) show how the method can be applied to the analysis of the productivity of alternative categories of rice germplasm in India.

A second somewhat more indirect hedonic approach to measuring the global value of plant breeding efforts by members of the CGIAR group is simply to map the geographical flows of improved varieties as an explanatory variable of productivity increases experienced in recipient countries over the introductory period. Evenson and Gollin (1997) use this method to assign a value to genetic improvements undertaken by the International Rice Research Institute. As they point out, the main drawback of this approach is that instead

[14] Consultative Group on International Agricultural Research.

of measuring the direct relationship between germplasm and productivity gains, the approach is indirect in measuring the tendency of germplasm collections to induce changes in the rate at which varieties flow across countries.

7.10 Towards economic valuation protocols

In some countries efforts are being made to standardise procedures for estimating economic values for environmental damage. In Canada, for example, ‘protocols’ are being researched and developed for use in assessing environmental damages that may be relevant for compensation and liability purposes. MacDonald *et al.*, (2000) describe progress on such protocols as they relate to pollution incidents in the Canadian Atlantic region. Of particular note is the existence, since 1995, of The Environmental Damages Fund, a trust account of Environment Canada. The Fund receives the proceeds from fines, court awards, out of court settlements etc. and the revenues are then used to support restoration and remediation processes. Economic valuation techniques are used to help determine the appropriate levels of damage. Environmental damage assessment (EDA) is currently limited because of resource limitations and the need to process incidents quickly. An EDA ‘Field Guide’ is being planned. MacDonald *et al*., (2000) give an example of a fish kill due to chemically contaminated soil being deposited in a river after heavy rainfall. The resulting damages were assessed as follows:

a. the investment costs of replacing the lost trout were estimated at around CAD$ 10,000 in the first year, an example of using replacement costs;

b. the costs of the investigation and enforcement of the ruling were debited to the event, at around CAD$ 15,000;

c. since the fishery was closed for recreational purposes to the local community, ‘standard’ consumer surplus measures were applied. Depending on the speed of reopening the fishery, losses were estimated to be in the tens of thousands of dollars. This illustrates the benefits transfer procedure (Chapter IX).

The Canadian study suggests that procedures for damage assessment can be streamlined through the development of protocols, and that ‘ready made’ valuation procedures can be applied in non-complex cases. For larger incidents, the estimates of consumer surplus loss are likely to become more important as the changes will be non-marginal.

VIII. STATED PREFERENCE METHODS

8.1 Introduction

The valuation methods in the previous section relied on existing markets to identify the environmental value of interest. The value is in some sense complementary to a market good such as housing or travel costs. For many environmental goods, no such 'behavioural trail' exists and a market must be constructed using questionnaires. This is the essence of stated preference (SP) methods. Drawing on advances in market research and cognitive psychology, the stated preference method has been applied widely in environmental economics over the last two decades.

An important feature of SP methods is that they can help reveal values that are not revealed using other methods. In particular SP can uncover non-use values (also known as passive use values). These are values reflecting a willingness to pay (WTP) for a good even though the respondent does not currently use it directly, nor intend to use it in the future. By definition, market values tend to reflect actual use and hence ignore non-use values. So long as non-use values are judged to be a 'proper' value for inclusion in a cost-benefit analysis, SP techniques are therefore capable of being more comprehensive than revealed preference techniques. Non-use values tend to be important in certain contexts, notably when the good in question has few substitutes. Since many biological resources are by definition unique, their non-use value is likely to be significant.

SP techniques use questionnaires that are targeted at a sample of individuals and that seek to elicit, directly or indirectly, the individuals' monetary valuation of a change in a given non-market good. Direct elicitation involves questions that take the form 'what are you willing to pay?' or 'are you willing to pay X?'. Indirect elicitation involves seeking rankings or rating by individuals across alternative options, each of which has some set of attributes or characteristics. One of the characteristics will be a price. Others might be distance that needs to be travelled to secure the good and some quality feature of the good. Careful analysis of the answers enables the relevant willingness to pay to be inferred, rather than have the respondent state their WTP. The direct elicitation procedures are defined as contingent valuation, and the indirect procedures are defined as attribute-based choice modelling. Both contingent valuation and choice modelling estimate the total economic value of the change in the non-market good. The methods have much in common in terms of question format. Both can, in principle, estimate the individual WTPs for the characteristics of the change in the good. In practice, choice modelling is better suited to the measurement of individual characteristics.

There are several distinct ways in which contingent valuation questions can be designed, and there are several distinct ways in which alternative choice options can be presented to individuals in choice modelling. All SP techniques share a common structure:

a. There must be a careful delineation of what it is that is being valued, i.e. what is the good in question and what is the nature of the change in the provision of that good? This information combines to produce a *scenario* and it is this scenario that respondents value. Several scenarios may be presented but care has to be taken not to 'overload' respondents so that they become confused about what it is they are being asked to value.

b. Information and data about the good must be collected that help describe the good and the way in which its provision will change, for example photographs could be used.

c. There must be a sampling strategy which will be either probabilistic, for example a statistically random survey, or non-probabilistic, for example, selecting respondents by some form of judgement.

d. A choice has to be made between the potential types of survey: by telephone, by mail and by face-to-face interview or mixed surveys such as mail and telephone.

e. Information and data about the respondents must be collected. This information will typically comprise:

   – socio-economic characteristics of the respondents (e.g. age, education, income);

   – attitudinal characteristics of the respondents;

– rankings, ratings (for choice modelling) or WTP responses (for contingent valuation) from the respondents.

8.2 The contingent valuation method

The contingent valuation method (CVM) is a survey technique[15] that attempts direct elicitation of individuals' (or households') preferences for a good or service. It does this by asking the respondents in the survey a question or a series of questions about how much they value the good or service. People are asked directly to state or reveal what they are willing to pay in order to gain or avoid some change in provision of a good or service. Alternatively, they may be asked what they are willing to accept, to forego or tolerate a change.

A contingent market defines the good itself, the institutional context in which it would be provided, and the way it would be financed. The situation the respondent is asked to value is hypothetical (hence, 'contingent'), although respondents are assumed to behave as if they were in a real market. Structured questions and various forms of 'bidding game' can be devised involving yes/no answers to questions regarding maximum willingness to pay. Econometric techniques are then used on the survey results to find the mean bid values of willingness to pay.

Over the last two decades interest in CVM has increased for a number of reasons (see Carson, 2000). First, a stated preference approach is the only means available for valuing 'non-use' (or passive use) values, such as people's existence values for a unique natural habitat or wilderness area. Second, the evidence available suggests that estimates obtained from careful and well-designed, properly executed surveys appear to be as good as estimates obtained from other methods. Thirdly, the design, analysis and interpretation of surveys have improved greatly with advances in scientific sampling theory, benefit estimation theory, computerised data management and public opinion polling.

Contingent valuation has been used extensively in the valuation of biological resources including rare and endangered species, habitats and landscapes. Limitations arise in terms of the information provision requirements necessary to allow respondents to value complex processes or unfamiliar species or ecosystem functions. Nevertheless, the method has been

[15] For a detailed review of the Contingent Valuation Method, see Mitchell and Carson (1989) and Carson (2000).

extremely useful in ex ante assessments of conservation policy and in ex post policy evaluation relating to conservation (Willis *et al.,* 1996).

8.3 Design of a CV study

In designing a CVM study, one has to answer a number of questions relevant to contingent valuation research in general. These include:

- What change in environmental quality should respondents be asked to value, and how should this change be described to them?

- What type of interview format should be used in the survey (i.e. face to face, telephone, or mail)?

- What type of questions (elicitation procedure) should be used to elicit respondents' valuation of the change in environmental quality?

- Exactly how should respondents be told that they would have to pay for the change in environmental quality?

- How can we increase our confidence that respondents in the contingent valuation survey are actually valuing the specific change in environmental quality described and not some other environmental quality change, and furthermore, that the values found are correct?

Some of the issues are described in more detail:

1. The Change in Environmental Quality that Respondents Are Asked to Value. A contingent valuation exercise requires that a hypothetical description (scenario) of the terms under which the good or service is to be offered is communicated to the respondent. Information is provided on the quality and reliability of provision, its timing and provision mechanism. Background information should also be provided on the various functions of the good or environmental service, and its importance to the economy and people of the affected region. This information should be sufficiently detailed to provide a common context for evaluating the environmental change offered in the questionnaire, but non-technical enough that it could be understood by the general public.

2. Interview Format. Interviews for a CVM study can be conducted by mail, telephone, face to face, or some combination of these. Mail and telephone surveys have been used frequently in the past. These methods are less expensive than face-to-face interviews, and avoiding biases caused by the quality of the enumerator. However, both methods introduce their own biases and are considered inferior to a face-to-face interview. Face to face interviews generally provide the best quality data, so long as interviewers have been well-trained. Their main problem as noted above is that they are expensive and can suffer from enumerator-induced biases.

3. Elicitation Procedure. The respondent is asked questions to determine how much s/he would value a good or service if confronted with the opportunity to obtain it under the specified terms and conditions. Respondents are often reminded of the need to make compensating adjustments in other types of expenditure to accommodate this additional financial transaction. The respondent’s choice or preference can be elicited in a number of ways. The simplest is to ask the respondent a direct question about how much s/he would be willing to pay for the good or service - known as continuous or open-ended questions. High rates of non-responses and zero responses can be a problem with this approach. Alternatively, a respondent can be asked whether or not s/he would want to purchase the service if it cost a specified amount. These are known as discrete, dichotomous choice, or referendum questions, and receive favour because they give the respondent no incentive not to answer truthfully, that is, the approach is incentive compatible. The discrete choice approach can be extended to have multiple bounds, although a double-bounded format has been found to have some efficient properties in terms of minimising the tendency for the respondent to say yes continuously. Open-ended questions, as well as single and double-bounded closed-ended questions are now the most frequently used formats in contingent valuation. Payment cards and iterative bidding formats used to be popular but are less so now since they are thought to introduce specific biases.

4. The Payment Vehicle. In order for respondents to consider seriously the hypothetical good or service described and the choices they are being asked to make, such that the answers they provide are the same as their actual behaviour if offered a real choice, the respondents must consider the actual manner they would be asked to pay for the good or service. The method of payment, the institution responsible for collecting the payment, and the duration of payments are collectively known as the payment vehicle. The payment vehicle should be both fair and reliable, and the length of time over which respondents commit to pay should contribute to the believability of the hypothetical scenario.

5. Tests of the Reliability, Bias and Validity of the CVM. In order to assess the technical acceptability of CVM practitioners employ various methodological tests to judge the reliability and to evaluate bias and validity of responses. There is a large and growing literature offering empirical tests in each area. Reliability concerns the degree to which the variance of WTP responses can be attributed to random error. The greater is the degree of non-randomness, the less the reliability of the study, such that mean WTP answers are of little significance. The variance arises as a consequence of true random error, sampling procedure, and the questionnaire/interview itself. The first of these is essential to the statistical process, while the second can be minimised by ensuring a sufficiently large sample size. The third relates to the issue of bias, they are considered in turn.

Bias refers to non-randomness in the variance of valuation responses. Bias can be caused by a number of factors that introduce bias into respondent behaviour. Well-documented biases include:

- strategic bias - respondents deliberately misstate their WTP;
- payment vehicle bias - WTP varies with the instrument suggested for payment;
- hypothetical bias - WTP is over or understated relative to what would be paid in a real market;
- starting point bias - WTP is ‘anchored’ on the first suggested bid price;
- insensitivity to scope - WTP is not affected by the scale of the good being offered;
- aggregation bias - aggregate WTP is sensitive to the number of people over which WTPs are aggregated.

The literature details several tests of the validity of contingent valuation studies. The most important are criterion validity and construct validity. The latter checks whether the results of CV studies are in line with theoretical expectations. The former tries to compare CV markets with behaviour in real markets.

Box 8.1 provides an example of CVM in Denmark.

Box 8.1 Economic Valuation of Recreation in Denmark

Although presented as a recreational survey, the Mols Bjerge study (Dubgaard *et al*, 1996), one of the earliest Danish CVM studies conducted in Eastern Jutland, is actually more concerned with the value of a semi–traditional managed heathland (2,500 hectares of managed grassland and woodland) that is actually a complex agricultural system relying on marginal grazing. The fact that the system is partially dependent on agricultural grazing raised the question of how it would be maintained given the falling viability of marginal agricultural systems. Against a culture of guaranteed free access to recreational areas, the use of a contingent valuation survey sought to estimate the economic value that visitors would place on the maintenance of the landscape. The main advantage of CVM is its ability to measure non-use value. That is, it can be used amongst a sample of non-users. The method can be said to measure the total economic value held by respondents irrespective of their use experience of the resource. This study restricted its focus to a sample of users (3,300) who were asked their willingness to pay (WTP) for a one-year entrance pass. Sensitivities related to historical recreational access rights meant that respondents might legitimately claim their property right and reject the notion of having to pay to safeguard access. In more technical terms, the status quo implied that a question asking their willingness to accept payment to avoid losing access might be the technically correct format in this case. If so, the question format would surely introduce biases in inflated compensation demands.

In the event, respondents did not protest against the WTP format. Open-ended and closed-ended question formats were used for the WTP question. Either format requires the collection of a number of explanatory socio-economic characteristics to check the validity of WTP statements. The mean WTP figure was derived using both methods and a common disparity between open-ended and closed-ended (typically higher) was discovered. The two mean WTP figures were used to calculate the aggregate WTP per year, multiplying each by the total population of users annually (130,000 people). This provided a range of total value that could then be discounted to provide a capitalised recreational value on a per hectare basis if necessary. Importantly this value was shown to be far in excess of the prevailing agricultural land price.

This study provides some noteworthy points for the use of the CVM. From a methodological point of view the choice of user rather than a non-user population circumvents a problematic choice of deciding arbitrarily the extent of the aggregate population that has a stake in the change. Aggregation over a potentially enormous population of non-users has frequently dented the credibility of CVM.From a policy perspective the study demonstrates how a sophisticated method can be adopted and pragmatically modified to evaluate the net social benefit gained from supporting conservation in Mols Bjerge..

8.4 Analysing CVM data

There are three ways in which CVM information is typically analysed to check the consistency of responses and to calculate the required valuation estimates.

First, summary statistics, such as means and median WTP can be used to calculate estimates of a good's total value for a particular population. Valuation frequency distributions can be used to estimate the proportion of a population that would be prepared to pay a given amount for the good.

Secondly, cross tabulations between WTP and socio-economic and other variables are considered. Questions on such socio-economic characteristics of respondents must thus be asked in the survey questionnaire. When point estimates of WTP are available, mean WTP bids can be calculated for different groups of respondents, and then checked against the predictions of demand theory. Cross tabulations for dichotomous choice questions (Are you WTP $X? Yes or No) are also possible but require large sample sizes to permit sufficiently powerful tests of differences between groups.

Thirdly, multivariate statistical techniques are used to estimate a valuation function that relates respondents' answers to hypothesised determinants of WTP, such as socio-economic variables and the prices of substitute good and services. Open-ended question formats typically yield an arithmetic measure of central tendency for the responses (mean or median). The format provides a continuous explanatory variable (WTP) that can be regressed on explanatory variables such as income, age and say, recreational participation. The explanatory power of such models is typically low, although the procedure is usually only a check to see that the signs on the dependent variables correspond with a priori expectations. The resulting function can be used to predict the actual amount that an individual with particular characteristics would be prepared to pay. A closed-ended question provides a qualitative (yes/no) dependent variable that must be regressed on the amount the individual was asked to consider plus other explanatory variables similar to those used in the open-ended format. Closed-ended formats have become more popular in recent times and the sophistication of statistical analysis of discrete choice data has increased significantly (see Hanemann and Kanninen, 1999). Specifically, the models employed to analyse such data draw on random utility theory to reflect the fact that the WTP decision has a random element. Modelling discrete choice therefore involves decisions about the form of the utility function used to describe this decision, the shape of the random element of the choice, and the actual amounts offered to respondents to consider. The simplest information format for combining this information and analysing simple qualitative choice data is a logit model. It can use Maximum Likelihood estimation of the function that relates the probability of being willing to pay an offered amount. The model choices have a direct influence on the derivation of the mean or median WTP value.

Table 8.1 The NOAA guidelines for contingent valuation

NOAA GUIDELINES
• Use personal interviews
• Use a WTP measure rather than WTA
• Use a dichotomous choice format
• Adequately pre-test the survey instrument
• Carefully pre-test any photographs used
• Use an accurate scenario description
• Favour a conservative design (more likely to under rather than over-estimate WTP)
• Deflect warm glows (overstatement of WTP to appear generous)
• Check temporal consistency of results
• Use a representative sample (rather than a convenience sample)
• Remind respondents of undamaged substitutes
• Remind respondents of budget constraints
• Provide a no answer or don’t know options
• Include follow-up questions to the valuation question
• Cross-tabulate the results
• Check respondents understanding

8.5 CVM and biodiversity valuation

The US National Oceanic and Atmospheric Administration (NOAA) panel has offered a set of guidelines on the CVM process. The recommendation was that these should be followed if CVM is to provide information about non-use values of sufficient quality as to be usable as the basis for claims for legal compensation for environmental damage (Arrow *et al.*, 1993). The use of these guidelines (Table 8.1) within the profession is now being extended to cover all CVM studies, although debate continues on the correctness of some of the guidelines. CVM is likely to be most reliable for valuing environmental gains, particularly when familiar goods are considered, such as local recreational amenities. A growing body of applications to biodiversity has been more or less specific about the subject of value (Loomis and White, 1996). The most reliable studies (i.e. those that have passed the most stringent validity tests and avoided severe 'embedding' whereby values are not sensitive to the quantity of the good being offered) appear to have been those that have valued high profile species or elements that are familiar to respondents. In other cases, the need to provide information to elicit reliable values is a limit to both CVM and other attribute based choice models.

8.6 Attribute based choice modelling

The term Attribute Based Choice Modelling (ABCM), or simply, Choice Modelling, encompasses a range of stated preference techniques that take a similar approach to valuing environmental goods. The term includes Choice experiments, Contingent ranking, Contingent rating, and the method of Paired Comparisons.

These methods are also sometimes known as "conjoint analysis", but this is a rather confusing term and one that should strictly be reserved for a technique of marketing for the launch of a new product. However, conjoint analysis has itself also been used for environmental valuation (Farber and Griner, 2000). The elements of ABCM that are common with contingent valuation are that the attribute scenarios are hypothetical choice sets. The questionnaire formats are also broadly similar. The differences are that the ABCM variants can be far more complicated to administer and, crucially, that the WTP is only elicited indirectly through a process of observed trade-offs made by respondents. Thus, whereas CVM directly asks for WTP, ABCM infers WTP from rankings or ratings of choice sets.

Choice modelling approaches are based around the idea that any good can be described in terms of its attributes and the levels that these take. For example, a forest can be described in terms of its species diversity, age structure, recreation facilities and an entry price or transport cost. Changing attribute levels will essentially result in a different "good" being produced, and it is on the value of such changes in attributes that choice modelling focuses. By choosing over these different "goods" including the implicit price attribute, respondents reveal the value of the other attributes indirectly.

Choice modelling conveys four pieces of information that may be of use in a policy context:

- The attributes that are significant determinants of the values people place on non-market goods.
- The implied ranking of these attributes amongst the relevant population(s). For example, in forests how broadleaf trees are ranked relative to conifers and how these are both ranked relative to improved access.
- The value of changing more than one of the attributes at once (for instance, if a management plan results in a given increase in wildlife protection but reduction in recreation access).
- As an extension of this, the total economic value of a resource or good.

However, it is important to stress here that not all ABCM approaches are equal in this respect. In fact, only two of them (choice experiments and contingent ranking) have demonstrably close links with economic theory, which allows the results to be interpreted as being equal to marginal (or total) values for use in CBA or in other contexts (Louviere *et al.,* 2000).

8.7 Choice experiments

In a choice experiment (CE) respondents are presented with a series of alternatives and asked to choose their most preferred. A baseline alternative, corresponding to the status quo situation, is usually included in each choice set. Each respondent is asked a number of these questions. The choice questions may also vary across respondents. Box 8.2 provides an example of choice modelling in Australia.

Box 8.2 Choice modelling in the Macquarie Valley, Australia

The competing uses of water often contrast productive or market uses such as agricultural production with the supply of non-market benefits associated with the maintenance of hydrological cycles in riverine and wetland systems and natural habitat for wild species. Weighing up the competing economic social and environmental values is a classic policy dilemma that can be informed by examining the economic trade-offs implicit in having more agriculture (e.g. greater yield productivity or employment), versus lower environmental quality (e.g. greater frequency of low flow in rivers and the lower incidence of species of fish or bird). Choice experiments allow these trade-offs value to be revealed and quantified in monetary terms. Bennet *et al.*, (2000) describe an application to the Macquarie Wetland System in New South Wales. Funded by the NSW Environment Protection Authority and the Natural Parks and Wildlife Service, the study seeks to explore the trade-off values of a sample of non-users for a bundle of socio-economic and environmental attributes associated with wetland conservation and use. As is common in contingent ranking studies these attributes were presented to survey respondents in a choice matrix that presented a series of 3 options characterised by a number of levels associated with five key attributes at stake, such as that shown in Table 8.2. The WTP vehicle used in this study was an increase in water rates for households, respondents being told that a state body would have to use this surcharge to "buy out" farmers' irrigation rights.

- ❑ I would choose option 1
- ❑ I would choose option 2
- ❑ I would choose option 3
- ❑ I would not choose any of these options because I would prefer more water to be allocated for irrigation.

Sequences of three options are presented to each respondent to choose between, but each new set of three options shows a different configuration of attribute levels. Supposing the respondent is asked to do this five times, by observing the choices made by respondents it is possible to develop a rich data set of implicit prices. Statistical modelling can be used to identify the parameters of part-weights indicated by the choices. These weights provide the information to value a change in the availability of each attribute. In summary for the Macquarie case, the analysis showed that increasing the breeding frequency (of waterbirds) by 1 year = 154 jobs = 545 km² of extra wetland area = 5 extra endangered or protected species present.

Box 8.2 continued over page

Box 8.2 continued

This breakdown of information is the strength of the Choice Modelling approach relative to contingent valuation, which provides an aggregate WTP value but rarely any more detailed information on the values of specific part of a whole package. In policy terms, the specific tradeoffs are far more relevant. Beyond the attribute values Choice Modelling can also provide the implicit WTP value for any configuration of attributes that constitute a probable option that might result from a project or policy change.

Table 8.2 Example of Choice Set from the Macquarie Marshes Questionnaire

	Option 1: Continue current situation	Option 2: Increase water to Macquarie Marshes	Option 3: Increase water to Macquarie Marshes
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
Endangered and protected species present	12 species	25 species	15 species

8.8 Contingent ranking, rating and paired comparison methods

Contingent ranking is very similar in spirit to the form of a choice experiment, except here the respondent is asked to rate the proposed options in Table 8.2 from most (1) to least (3) favoured according to their preferences. In practice, more than three options are usually specified. Contingent rating proposes a very similar exercise to that in a ranking exercise. In a rating exercise respondents are presented with a number of scenarios and are asked to rate them individually on a semantic or numeric scale. This approach does not therefore involve a direct comparison of alternative choices. Instead, respondents are asked a series of such questions, where the policy design varies.

In this way, data are collected on rating scores for different 'designs' of the environmental good or policy.

The example in Table 8.3 considers such an approach for rating an option for landscape and habitat conservation. An extension to this approach is a paired comparison exercise. Here respondents are asked to choose their preferred alternative out of a set of two choices and to indicate the strength of their preference on a numeric or semantic scale. This approach combines elements of choice experiments (choosing the most preferred alternative) and rating exercises (rating the strength of preference).

Table 8.3 Illustrative contingent rating question

On the scale below, please show how strongly you would prefer the following policy option

Characteristics	Option 1
Native woodland	500 ha protected
Heather moorland	1200 ha protected
Lowland hay meadow	200 ha protected
Cost per household per year in additional taxes	£25

Please tick one box only

1	2	3	4	5	6	7	8	9	10

Very low preference — Very high preference

8.9 Common design features

All the above stated preference techniques share common design features. There are some similarities to the contingent valuation process, but with key differences in a design process that allows WTP to be elicited indirectly. There are five important design stages:

Stage 1: Identification of relevant attributes to include in choice sets.
Stage 2: Assignment of levels to attributes including the price attribute.
Stage 3: Determining the factorial design set of matrices combining attributes and levels to be presented to respondents.

Stage 4: Determining an efficient subset of the matrices to present to a sample of respondents.

Stage 5: Administering the survey in a face-to-face format.

8.10 Analysing ABCM data

The design and analysis of ABCM surveys is complicated relative to CVM. However, the design and analysis draws on the random utility framework that characterises the models used to analyse closed-ended CVM. In the design stage, the main challenge arises in determining the essential attributes that define the good to be valued and their appropriate levels. Because price is one of the attributes, this problem are similar to the bid vector design problem encountered in closed-ended CVM. The marketing literature provides guidance on the number of attributes and levels that is psychologically acceptable for the average respondent. Design algorithms can then aid in the task of reducing a complex factorial design down to the smallest set of combinations - depending on whether main effects are of interest or both main effects and interactions. The final combination of options then typically requires the individual respondent to rank or choose between sets of options (made up of attributes at varying levels) such that multiple choice observations are made for each individual. To model this information some assumption is necessary about the form of the indirect utility function that underlies the choice decision. Ordered logit models are then normally used to estimate the parameters of the choice from which marginal rates of substitution can be calculated.

8.11 Choice modelling versus contingent valuation?

ABCM places the respondent in much the same situation as a CVM survey. The key difference is that the cognitive process is somewhat circumscribed by the attributes (and levels) a respondent must choose between. For some goods, this may prevent the respondent making default assumptions. However, this depends on the amount of background information provided in what is already (for the respondent) a cognitively burdensome task. There is a small body of studies testing whether there is applicability of ABCM to biological resources. It can be argued that the constrained attribute design requirement of ABCM is even more limiting than CVM. Moreover, the selection and representation of these attributes and their levels simply adds to the design problems already associated with hypothetical surveys. A strong advantage of the ABCM over CVM is that the method can reveal something about the sum of the parts of a resource rather than the total value. In many circumstances, the policy question to be answered by a valuation study concerns the improvement of a specific attribute.

If resources permit, both contingent valuation and choice modelling studies can be undertaken to permit some form of convergent validity. Often, however, resources do not stretch to more than one technique and a choice has to be made. At present there is no strong reason to choose one technique in preference of the other. No case has been made to disprove the charge that the recent trend towards ABCM simply adds to the problems encountered using hypothetical surveys for no obvious advantage.

IX. ECONOMIC VALUATION: BENEFITS TRANSFER

9.1 The aim and nature of benefits transfer

Benefits transfer involves ‘borrowing’ an estimate of willingness to pay from one site (the study site) and applying it to another (the policy site). What is borrowed may be a mean value which is not adjusted or a mean value which is modified to ‘suit’ the new site. Or it may be a whole *benefit function* that is transferred. The attraction of benefits transfer is that it avoids the cost of engaging in ‘primary’ studies whereby WTP is estimated with one or more of the techniques described in Chapters VII-VIII. Moreover, valuable time can be saved if the benefits transfer approach can be used. The essential problem with benefits transfer, however, is its reliability. How can the transferred value be validated? The main procedure for validation involves transferring a value and then carrying out a primary study at the policy site as well. Ideally, the transferred value and the primary estimate should be similar. If this exercise is repeated until a significant sample of studies exists in which primary and transferred values are calculated for policy sites, then there would be a justification for assuming that transferred values could be used in the future without the need to validate them with primary studies. An alternative procedure is to conduct a *meta-analysis* on existing studies to explain why studies result in different mean (or median) estimates of WTP. At its simplest, a meta-analysis might take an average of existing estimates of WTP, provided the dispersion about the average is not found to be substantial, and use that average in policy site studies. Finally, benefits transfer could be tested by estimating WTP before an actual project is implemented and then revisiting the area later when the project is complete to see if people behaved according to their stated WTP.

Since benefits transfer is fairly new, there are few areas of environmental policy that have been subjected to detailed assessments of the validity of transfer. Brouwer and Spannincks (1999) list the areas that have been subjected to testing as sports fishing, water quality improvements, reservoir-based recreation, green space, and white water rafting. There are also recent studies on valuing health improvements. Studies of the *validity* of benefits transfer need to be distinguished from studies where benefits transfer is used. The latter is very common, especially in the context of air pollution from electricity generating sources. Significantly, however, the validity of these transfers is rarely tested.

9.2 Forms of benefit transfer

<u>Transferring average WTP from a single study to another site which has no study</u>

One elementary procedure is to 'borrow' an estimate of WTP in context i (the study site) and apply it to context j (the policy site). The estimate may be left unadjusted, or it may be adjusted in some way. Transferring unadjusted estimates is clearly hazardous, although it is widely practised. Reasons for differences in average WTP across sites include:

- Differences in the socio-economic characteristics of the relevant populations.

- Differences in the physical characteristics of the study and policy site.

- Difference in the proposed change in provision between the sites of the good to be valued.

- Differences in the market conditions applying to the sites (for example variation in the availability of substitutes) (Bateman *et al.*, 1999).

As a general rule, there is little evidence that the conditions for accepting unadjusted value transfer hold in practice. Hence, some form of adjustment should be made. Bateman *et al.*, (1999) distinguish various forms of adjusted transfer:

a. Expert judgement, i.e. experts make a judgement about how the WTP will vary between the study site and the policy site.

b. Re-analysis of existing study samples to identify sub-samples of data suitable for transfer.

c. ‘Meta-analyses’ of numbers of previous estimates permitting the estimation of cross study benefit functions applicable to policy sites.

A widely used formula for adjusted transfer is:

$$WTP_j = WTP_i\ (Yj/Yi)^e$$

where Y is income per capita, WTP is willingness to pay, and ‘e’ is the “income elasticity of WTP”, i.e. an estimate of how the WTP for the environmental attribute in question varies with changes in income. In this case, the feature that is changed between the two sites is income perhaps because it is thought that this is the most important factor resulting in changes in WTP. But it should also be possible to make a similar adjustment for, say, changes in age structure between the two sites, changes in population density, and so on. Making multiple changes of this kind amounts to transferring benefit functions (see below).

Transferring benefit functions: meta-analysis

A more sophisticated approach is to transfer the *benefit function* from i and apply it to j. Thus if it is known that $WTP_i = f(A,B,C,Y)$ where A,B,C are factors affecting WTP at site i, then WTP_j can be estimated using the coefficients from this equation but using the values of A,B,C, Y at site j. An alternative is to use *meta-analysis* to take the results from a number of studies and analyse them in such a way that the variations in WTP found in those studies can be explained. This should enable better transfer of values since we can find out what WTP depends on. In the meta-analysis case, whole functions are transferred rather than average values, but the functions do not come from the single site i, but from a collections of studies. In either case, the effect is to derive from the study site or sites, a WTP function. This might have a simple linear form, say

$$WTP = aA + bB + cC$$

where A, B, C are determining factors such as income etc. This function can be transferred to the policy site so that

$$WTP' = aA' + bB' + cC'$$

where WTP' is the WTP to be estimated, and A', B' and C' are the values of the variables at the policy site. Effectively, what gets transferred are the coefficients from the WTP equation derived from the study site(s).

9.3 Case study: a meta-analysis of UK woodland recreation values

Table 9.1 presents summary details from some 30 studies of UK woodland recreation value, yielding over 100 benefit estimates, reported in Bateman *et al*., (1999).

Table 9.1 Studies of UK woodland recreation value

Value type	Recreation value unit	Value method	No. of studies	Date conducted[1]	No. of value estimates	Value range (UK£, 1990) (m = million)
Use	Per person per visit.	CVM	8	1987–1993	28	£0.28 - £1.55
Use + option	Per person per visit.	CVM	3	1988–1992	16	£0.51 - £1.46
Use	Per person per visit.	ZTC	3	1976–1988	18	£1.30 - £3.91
Use	Per person per visit.	ITC	3	1988–1993	16	£0.07 - £2.74
Use	Per person per year	CVM	3	1989–1992	7	£5.14 - £29.59
Use	Per household capital[2]	CVM	3	1990	3	£3.27 - £12.89
Use	FC forests/ conservancy	TCM	1	1970	13	£0.1m - £1.1m
Use	Total UK value	TCM	6	1970–1998	6	£6.5m - £62.5m
-	All studies	-	30	1970-1998	108	-

Notes:
1 = all dates refer to the year of study survey rather than publication date.
2 = These studies use a once-and-for-all WTP per household question.
TCM = travel cost, CVM = contingent value method, ZTC = zonal travel cost method, ITC = individual travel cost method.

The aim is to determine what factors influence the resulting WTP estimates, where the factors in question include the methodology used in the study, the year the study was conducted, and the authors. Note that meta-analysis includes features reflecting how the study was conducted as well as features of the study content.

Linear regression equations fitted the data well. The strongest influence upon WTP results within the selection of CVM studies is whether use value alone or use plus option value is elicited; the latter providing values which are on average some 16% higher than the former. Two elicitation methods (methods for eliciting WTP, e.g. open-ended questions) produce estimates which differ significantly from others in the dataset. Open-ended questions produced values substantially higher values, and payment cards with wide ranges also produced high values. The *Year* variable proved insignificant. However two of the *Author* categories were significant, suggesting that who actually carried out the studies might have influenced the results. However the *Author* variable is clearly unsuitable for the purposes of benefits transfer.

A modified regression with 'authors' deleted, produced results with about 60% of total variation explained. The linear functional form allows straightforward interpretation of the coefficients as WTP sums. For example, the open ended elicitation approach produces an estimate of woodland recreation use value (excluding option value) of £0.60 per person per visit. The results from the modified equation were used to value estimates of visitor arrivals for the whole of Wales derived from models of predicted arrivals developed using geographical information systems (GIS). The outcome is a map of recreational value per area according to location over the whole of Wales.

The study was then expanded to include those based on the travel cost method, thus expanding the sample of studies going into the meta-analysis. Which methodology is used, i.e. contingent valuation, and specific type of travel cost method, now enters the equation as an explanatory variable. The valuation methodology used was found to affect the WTP estimate. Some variants of the travel cost method produced values which, *ceteris paribus*, were substantially higher than those produced by the other methods, such as CVM and individual travel cost procedures using different econometric techniques.

The addition of the travel cost studies to the contingent valuation studies in the meta-analysis increased the overall statistical fit from roughly 60% to 70%. Some features of the meta-analysis confirmed theoretical expectations: for example, use plus option value should be higher than use value alone, and this was confirmed in the meta-analysis. In other respects, however, the approach revealed some high risks in transferring estimates. From a transfer point of view, it is not desirable that estimates should be sensitive to the valuation methodology adopted, and even less satisfactory that values should be influenced by the author(s) of the original studies. The study therefore suggests considerable caution in adopting benefits transfer techniques at this stage.

9.4 Case study: a meta-analysis of wetland values

Brouwer *et al*., (1999) conducts a meta-analysis of wetland values. Thirty studies were finally used, producing over 100 estimates of WTP. In order to make the money valuation amount compatible between survey dates and across national boundaries all national currencies were expressed in terms of their countries' 1990 purchasing power expressed in units of Special Drawing Rights (SDRs). Average WTP across all studies was 62 SDRs while the median was considerably lower, at 34 SDRs. Table 9.2 illustrates how the original WTP estimates varied. Statistical regressions were run relating WTP to factors judged to be potentially influential on WTP.

The estimated coefficients in the semi-log function represent the constant proportional rate of change in the dependent variable per unit change in the independent variables. Hence, the coefficient estimated for the dummy variable 'payment vehicle' in the basic model reflects, *ceteris paribus*, an almost twice as high average WTP for an increase in income tax than for any other payment vehicle. It is possible that people were willing to pay more via income tax than via other payment vehicles. This is because the tax indicated the social relevance of the problem. But the general understanding that most people will pay, thus avoiding possible feelings of unfairness or injustice and hence avoiding lower bid amounts or even protest bids, could also have been relevant. It might also indicate the greater certainty or trust placed in this payment vehicle as an indication that the promised environmental services will actually be provided. The open-ended elicitation format is seen to produce a significantly lower WTP, by about 40%, than other formats. This may be due to the uncertainty experienced in answering an unfamiliar question in an open-ended format. The dichotomous choice format yields the highest average WTP, followed by the iterative bidding procedure.

The basic model also indicates that study location has a significant impact on average WTP. The dummy variable has a value of 1 if the research took place in North America and zero if in Europe. Average WTP appears to be substantially higher in North America than in Europe. Conversely, higher response rates, a rough indicator of overall study quality, appear to result in significantly lower average WTP than low response rates. Although Table 9.2

suggests that WTP increases overall with increasing relative wetland size, the statistical analysis does not bear this presumption out.

Table 9.2 WTP estimates in wetland studies (SDR, 1990)

	Mean (SDRs)	Standard Error	No. of Observations
Value Type			
use value	68.1	8.4	50
non-use value	35.5	4.8	13
use and non-use values	63.8	12.9	40
Wetland Function			
flood control	92.6	24.4	5
water generation	21.5	6.8	9
water quality	52.5	5.9	43
biodiversity	76.1	12.8	46
Relative Wetland Size			
very large	86.9	17.6	8
large	70.3	21.6	16
medium	67.0	8.9	58
small	29.5	13.2	13
very small	53.4	13.8	6
Country			
USA and Canada	70.8	7.8	80
Europe	32.8	8.4	23
Payment Mode			
income tax	121.3	18.1	22
private market[d]	28.6	5.7	28
product prices	47.8	8.9	22
combination of 1 and 3	42.8	6.3	26
trip expenditures	102.9	6.8	3
not specified	237.5	106.2	2
Elicitation Format			
open-ended	37.4	6.5	35
dichotomous choice	91.2	17.1	29
iterative bidding	78.5	14.9	20
payment card	47.1	8.4	19
Response Rate			
less than 30%	47.5	14.6	10
between 31 and 50%	46.9	9.2	25
more than 50%	78.3	9.9	59

Source: Brouwer *et al*., (1999).
[1] Private fund/Entrance fee.

Table 9.3 reports the variables which were statistically significant at the 10% level.

Table 9.3 WTP for wetlands: regression results for the basic and extended model

		Basic Model		Extended Model	
Parameter	Parameter Definition	Estimate	Standard Error	Estimate	Standard Error
Constant	Intercept	3.356***	0.100	3.311***	0.247
Payment vehicle	Dummy: 1 = income tax 0 = other	1.880***	0.265	1.576***	0.362
Elicitation format	Dummy: 1 = open-ended 0 = other	-0.411**	0.130	-0.376*	0.183
Country	Dummy: 1 = North America 0 = other	1.861***	0.217	1.629***	0.363
Response rate (1)	Dummy: 1 = 30-50 percent 0 = other	-2.253***	0.326	-1.722***	0.451
Response rate (2)	Dummy: 1 = > 50 percent 0 = other	-1.904***	0.333	-1.461**	0.450
Flood control	Dummy: 1 = flood control 0 = other	1.477***	0.240	1.134*	0.456
Water generation	Dummy: 1 = water generation 0 = other	0.691*	0.342	0.441	0.479
Water quality	Dummy: 1 = water quality 0 = other	0.545†	0.282	0.659*	0.327
Pseudo R-squared		0.365		0.380	
N		92		92	

† = significant at 0.10 ** = significant at 0.01
* = significant at 0.05 *** = significant at 0.001

More interesting is the role played by the wetland functions themselves since they have a statistically significant role in explaining variance in average WTP. The average WTP is highest for flood control, followed by water generation and water quality and lowest for the function of supplying or supporting biodiversity. This probably suggests that, in the contexts of wetlands, use values dominate, and the role of wetlands in supporting biodiversity may either not be understood fully, or is not thought to be important.

The suitability of the meta-analysis for benefits transfer is again the subject of some cautionary remarks by Brouwer *et al*. But the authors suggest that if low variance reflects the quality of the estimate for purposes of benefit transfer, then studies using income taxation as a payment vehicle are better suited than other payment vehicles, and studies valuing wetland biodiversity are more reliably transferred than estimates of the value of wetlands in their capacity of generating water or maintaining water quality.

9.5 Case study: the Szigetköz wetland in Hungary

Hungary and the then Czechoslovakia agreed to construct a Danube barrage system in the late 1970s. Subsequently, there was a dispute about the impacts of the dam, and Hungary ceased construction. But other construction continued and 'Variant C' of the dam proceeded. The fairly immediate effect was damage to some of the local ecosystems. It was decided to attempt to estimate the environmental damage as part of the process of the continuing negotiations between Hungary and Slovakia. But resources did not permit a detailed original valuation study. Hence benefits transfer was used. A study by Szerényi *et al*., (2002) argued that an earlier study for Austria was applicable to the Hungarian case since it related to the Austrian's willingness to pay for the creation of a national park along the Danube. The justifications for the transfer were that (a) the site characteristics were very similar; (b) the issue was conservation which is applicable in both the study site and the policy site; (c) the cause of the degradation was similar - a hydropower plant in the Austrian case and the dam in the Hungarian case, and (d) economic conditions are similar, although incomes per capita vary significantly. The initial Austrian value of some 330 Austrian schillings per person per year was taken.

The initial value was then adjusted in various ways to account for (a) differences in per capita income, (b) differences in the areas in question, (c) differences in the rates of environmental degradation. Simple procedures were adopted, e.g. the Austrian WTP was multiplied by the ratio of Hungarian to Austrian GNP per capita. The result was a WTP in Hungary of HUF 1.6 per person per year. This was then aggregated across all Hungarians above 14 years of age and adjusted slightly for differences in area. The result was an aggregate WTP of some HUF 17 billion per year. Since this is for total degradation of the resource, a 20% degradation would be one-fifth of this. Assuming valuations stretch over an infinite period, then the present value of damage can be estimated for varying discount rates. For 'Variant C' the sums became HUF 168-252 billion for a discount rate of 2% and HUF 96-144 billion for a 3.5% rate.

The sums in question could be used as the basis for compensation claims, illustrating one of the uses of economic valuation procedures. The validity of the transfer method is, as noted in the text, open to question, but the Hungarian study was careful to compare the situation with a very similar one, and the transferred values were adjusted for the most obvious factors affecting WTP in Hungary. Additionally, the study tested the resulting transferred WTP against contingent valuation estimates for other ecological systems in Hungary. Probably the more debatable issue is the procedure of aggregating across all Hungarian people above the age of 14 years. There is mixed evidence on how WTP varies with distance from the site being valued, with some evidence suggesting that people further away will attach less value to the site that those nearer to it.

9.6 Testing for the validity of transferring benefit functions

The testing of benefit function transfers requires that the regression coefficient estimates are compared, and this involves selecting econometric tests (for detail see Bateman *et al.*, 1999). The aim is to see if the coefficients in the benefit equation are the same at the two sites. The issue can be illustrated with an example from the USA.

Loomis (1992) tested the transferability of three sets of multi-site travel cost functions as follows:

i. Transferring functions for sea fishing between Washington and Oregon state.
ii. Transferring functions for freshwater fishing between Oregon and Idaho state.
iii. Transferring functions for freshwater fishing within Oregon state.

The basic equation is:

$$T_{ij}/\mathrm{POP}_i = B_0 - B_1 TC_{ij} + B_2\mathrm{TIME}_{ij} + B_3\mathrm{SUBS}_{ik} + B_4\mathrm{INC}_i + B_5\mathrm{QUAL}_j$$

where

T_{ij}	trips from origin *i* to site *j*
POP_i	population of origin *i*
TC_{ij}	travel costs of origin *i* to visit site *j*
$TIME_{ij}$	travel time of origin *i* to visit site *j*
$SUBS_{ij}$	a measure of the cost and quality of substitute site *k* to origin *i*
INC_i	average income of origin *i*
$QUAL_j$	recreation quality at site *j*.

As noted earlier, successful benefit function transfer occurs when the coefficients at the study site (the B coefficients above)are equal to the 'true' coefficients at the policy site (say the 'A' coefficients) such that:

$B_0 = A_0$ and $B_1 = A_1$ and $B_2 = A_2$ and $B_3 = A_3$ and $B_4 = A_4$ and $B_5 = A_5$ and ... B_n

Loomis's approach can be illustrated for the transfer of functions for sea fishing between Washington and Oregon state. The simple demand specification for Oregon and Washington ocean sport salmon respectively are set out below:

$$\ln (T_{ij}/\mathrm{POP}_i) = \underset{(-0.93)}{-2.1285} - \underset{(-8.68)}{1.07} (\ln \mathrm{DIST}_{ij}) + \underset{(1.82)}{0.401} (\ln \mathrm{FISH}_j)$$

where:
$DIST_{ij}$ is roundtrip distance from origin *i* to port *j*
$FISH_{ij}$ is total sport harvest of salmon at port *j*

$F = 37.7$ ($p < 0.01$)
$R^2 = 0.62$.
(figures in brackets are t-values)

$$\ln (T_{ij}/\mathrm{POP}_i) = \underset{(-4.80)}{-5.643} - \underset{(-11.47)}{0.94} (\ln \mathrm{DIST}_{Dij}) + \underset{(7.07)}{0.592} (\ln \mathrm{FISH}_j)$$

$F = 128$ ($p< 0.01$)
$R^2 = 0.53$.

Econometric tests indicated that equality of coefficients must be rejected. Similar analysis of the transfer of functions for freshwater fishing between Oregon and Idaho state rejects equality of coefficients at the 0.05 significance level. Loomis suggests that these failures of transferability may in part be due to changes in angling behaviour or other differences occurring between the various survey dates underpinning this comparison. For the analysis of freshwater fishing within Oregon state, Loomis looks at ten sites and then uses an analysis of 9 of the sites to see if it will predict the value at the tenth site. This is now a common procedure in meta-analysis, i.e. for any N studies, the nth study is omitted from the meta-analysis and N-1 studies are used for the meta-analysis to see if they predict the results in the Nth case. Loomis's analysis suggests that the resulting transferable estimates are reasonable, i.e. they do not differ markedly from the actual values derived in the original study. Moreover, the transfer approach does produce a better approximation than that obtained by simply using the average values derived from the full 10 site model.

9.7 Conclusions

Benefits transfer is still a developing subject**.** In many respects it is the 'Holy Grail' of approaches based on economic valuation. Ultimately, it may be possible to borrow unit values from a library of valuation studies and apply them to new sites and issues. To date, however, the literature that tests for the validity of benefits transfer is a long way from supporting such procedures. It seems clear that factors such as the quality of the original studies may be very important in explaining why value estimates may differ, and this is not satisfactory for benefits transfer purposes. As more and more validity studies are carried out, the nature of the factors explaining differences in WTP across sites will become clearer. At the moment there appears to be no substitute for high quality original studies.

X. BIODIVERSITY VALUES AND THE POLICY PROCESS

10.1 The policy context for biodiversity values

The previous chapters have sketched out the nature of the values to be attached to biological resources and biodiversity and the ways in which these values might be elicited. The topic is clearly a wide ranging one since it embraces the fundamental notions of moral and aesthetic value, instrumental value, and the ecological and economic importance of biological systems. Perhaps the most fundamental feature of the policy context in which biodiversity values need to be deliberated is the fact of choice. Not all biological resources can be conserved, nor in an evolutionary context should they be conserved. But the current rates of extinction and habitat loss show that policies are operating in a context of non-evolutionary change. Hence choices need to be made about biodiversity conservation in the face of rapid biodiversity loss and with limited economic budgets.

Table 10.1 suggests the relevant policy contexts in which estimates of biodiversity values are needed. While linking policy contexts with values, the table does not attempt to be prescriptive about specific valuation approaches to be used. This is because the choice of approach is highly context specific and in most contexts several valuation methods could apply. Figure 10.1 complements the table by summarising the methodological options than can be pursued. The diagram makes a distinction between economic valuation and alternative criteria that may dictate conservation decisions. Non economic criteria determined by moral, cultural or spiritual values may become meta-criteria in cases where they are significant enough to override any economic consideration. In other cases the accommodation of these values may simply be pursued by means of cost-effectiveness analysis as a single criterion or as part of a multi-criteria analysis.

The process of 'demonstrating' the value of biodiversity - drawing it to the attention of the public and to decision-makers - is something that can make effective use of all the procedures discussed for 'valuing' diversity.

Table 10.1 The policy contexts of biodiversity values

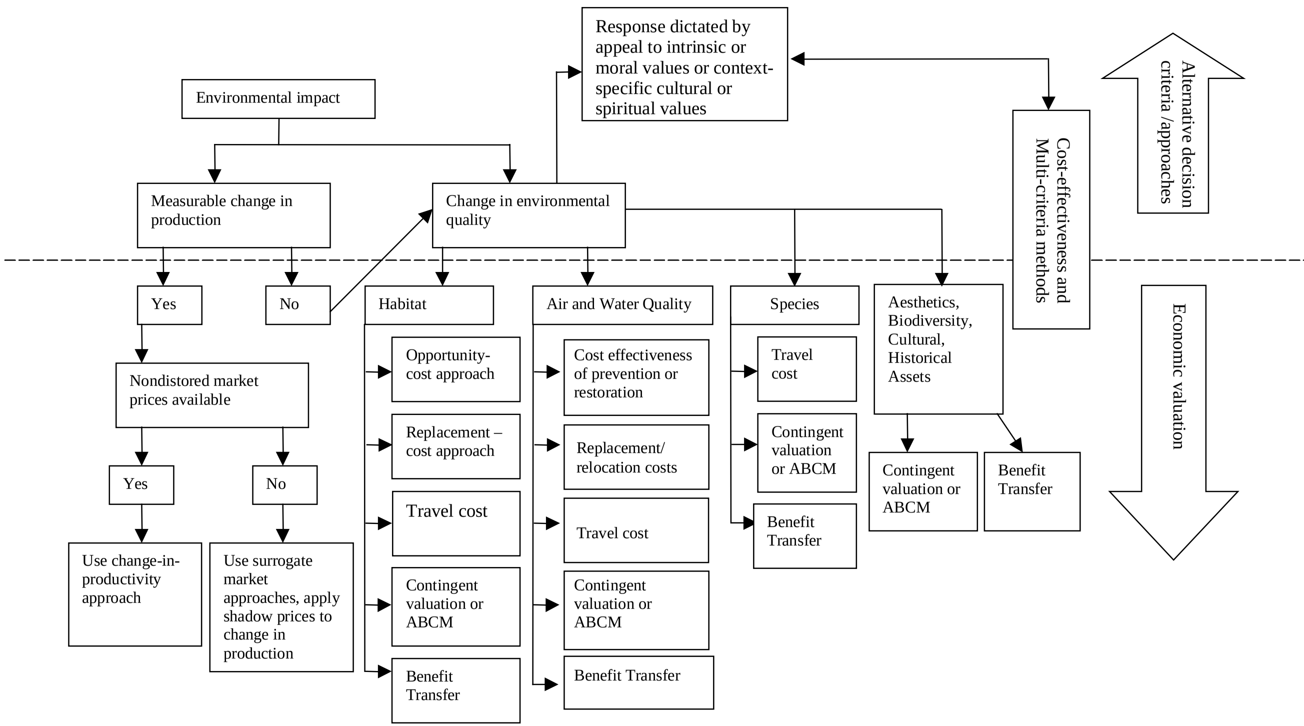
Context	Type of values
1. Demonstrating the value of biodiversity: awareness raising, showing the importance of biodiversity	All notions of value: moral, aesthetic, economic, ecological. Moral approaches could be based on ascription of 'rights' to sentient things, religious views, notions of stewardship. Aesthetic approaches would refer to the intrinsic qualities of beauty in biological resources. Ecological approaches would stress role of diversity in ecosystem resilience, provision of information, and role of existing diversity in future evolution. Economic approaches would stress role of diversity in resilience, in the provision of (especially) genetic information, and in the provision of ecosystem goods and services.
2. Determining damages for loss of biodiversity: liability regimes	Liability is usually assessed in monetary terms and relates to notions of compensation for loss, or to notions of restoration of pre-damage conditions. In former case, explicit monetary valuation is required and may or may not be based on economic valuation approaches. Restoration of damage approaches may be based on 'public trust' doctrines that do not consider cost (e.g. in the USA, Endangered Species Act, 2001 Supreme Court ruling on Air Quality standards, lower court rulings on liability regimes).
3. Revising the national economic accounts	Two approaches. Economic approaches are required for full national accounting in which damage to biological resources is deducted (along with other damages and resource depletion) from conventional Net National Product. Non-economic approaches, or 'satellite accounting', leave conventional national accounts as they are and add physical indicators of changes in the stock of biological resources. Possible to add changes in an index of diversity but not so far done.

Table 10.1 continued over page

Table 10.1 continued

4. Setting charges, taxes and fines	Economic valuation has been used to determine the size of charges and taxes (e.g. in UK, Landfill Tax and Aggregates Tax). Ecological indicators could be used to stratify taxes etc, e.g. higher taxes for ecological more important impacts.
5. Land use decisions: e.g. - encouraging sustainable agriculture - encouraging sustainable forestry - making case for protected areas	Multi-goal techniques, cost-effectiveness and cost-benefit all relevant. Involves a notion of cost of policy measure and some measure of effectiveness. Only CBA allows decision as to whether any policy is worthwhile.
6. Limiting biological invasions	Cost-effectiveness procedures: cost of measure needs to be compared to expected conservation of diversity.
7. Limiting or banning trade in endangered species	Economic approaches are relevant, but policies tend to be based on notions of absolute importance, plus limited social value of trade. Latter can be construed as a 'safe minimum standards' approach whereby conservation is favoured unless opportunity cost of conservation is 'high'.
8. Assessing biodiversity impacts of non-biodiversity investments, e.g. road building, airports, housing development	All measures of biodiversity are relevant and can be included in monetised or non-monetised form in integrated appraisals.
9. Setting priorities for biodiversity conservation within a limited biodiversity budget	Cost-effectiveness or cost-benefit procedures required.

Figure 10.1 A valuation flowchart



Source: Adapted from Dixon and Sherman 1990.
Note: ABCM = Attribute-based Choice Modelling.

More complex is the issue of determining the damages involved in liability; that is, legal regimes where an agent is held liable for biodiversity loss. Examples include losses from oil spill accidents where restoration costs, or costs imposed in terms of clean-up have been used. Economic valuation procedures, including those based on stated preference techniques, have also been used in such contexts, as in the *Exxon Valdez* oil spill in Alaska. In other cases, the 'public trust' doctrine has been invoked whereby liability damages are determined by whatever it costs to restore the environment to some pre-damage state. In still others, court proceedings have determined damages without explicit guidance from economic approaches.

There is now a substantial effort being made to adjust national income accounts to secure better representation of the true levels of economic activity and its effects within a nation. Modified national accounting takes two forms: one in which all environmental impacts are valued in monetary terms and used to adjust the net national product estimate for the economy[16], and the other in which conventional accounts are linked to physical indicators of environmental change ('satellite accounts'). Such procedures assist in making it clear that all economic activity feeds back into gains or losses in biodiversity, enabling better macroeconomic planning decisions to be made.

Land use change is the major cause of biodiversity loss, so that land use decisions are of considerable importance for biodiversity. Because of this, the land use issue is discussed in more detail in Section 10.2.

Two other significant causes of biodiversity loss arise from the trade in endangered species and biological invasions. The former depletes the biodiversity asset base in the 'exporting' country, reducing minimum viable populations, and the latter results in inter-species competition for space and food supplies. International agreements such as the Convention on International Trade in Endangered Species (CITES) have not typically adopted economic approaches to value. The assumption has been that any costs of banning or regulating trade are outweighed by the ecological value of the species in question. The value debate is central to the effectiveness of CITES since it has been argued that it diverts attention away from the probably more important causes of loss, namely habitat change. Also, CITES works on the basis of trade being 'wrong' without asking whether a controlled trade could act as a

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16 Essentially, this is done by treating any loss of biodiversity as a depreciation of an asset, and any gain as an appreciation. Conventional net national product (NNP)(GNP - depreciation on man-made assets) is then modified to adjusted NNP which is equal to NNP - the depreciation on biodiversity. Other assets can be treated in the same way.

mechanism for investing in biodiversity conservation. This debate has been most active in the context of African elephant conservation. On the one hand, those favouring the CITES approach have argued that the source of demand for illegal ivory should be controlled and suppressed, thus reducing the incentive to poach. On the other, those states that have successfully managed large elephant populations point out that elephants do considerable damage, reducing the incentive of local people to conserve them. Through sustainable culling, elephant products can be sold and the revenues reinvested in local communities (see Hutton and Dickson, 2000). This debate is very much one about issues of fact - sustainable use versus protection as the most effective means of conservation - and about issues of value - moral versus economic value.

The issue of biological invasions is also complex. Preventing new invasions is an issue of restricting the import of any species that will compete with endemic species. But the more difficult issue is whether to adjust existing non-endemic species which may have been present for hundreds of years and which still compete with indigenous species. Here some notion of costs and benefits is relevant, but biological invasions may also exhibit strong discontinuities with domestic species populations exhibiting sudden collapse (Perrings *et al.*, 2000).

Biodiversity is affected by everyday economic investment decisions such as infrastructure investment. Economic and physical measurement of biodiversity impacts can help in revising investment decisions that might otherwise ignore these impacts. Cost-benefit and multi-criteria approaches are the most relevant in this context.

Finally, the issue of priority setting was discussed in Chapter II. It was shown there that simply identifying the location of the greatest diversity is not sufficient to determine sound conservation policy. Additional features are the degree of threat, the cost of the intervention and the probability of being successful. Section 10.4 below expands further on these issues.

10.2 Land use decisions and sustainable use of biodiversity

Because land use change is important as a cause of biodiversity loss, it is treated in a little more detail in this section. Land use change tends to be prompted by economics, or rather, financial comparisons of rates of return to different land uses. If the rate of return to some developmental use, say agriculture or residential land, is r_a, and the rate of return to conservation is r_c, then the likelihood is that the land will be converted from a use consistent with conservation if:

$$r_a > r_c \tag{1}$$

On the economic model, this change of use would be justified if, in each case, the rates of return reflected the full costs and benefits of those particular land uses. But there are good reasons for supposing that the two rates of return are not ‘true’ returns.

First, some land uses are subsidised. Effectively, then, inequality (1) appears as:

$$[r_{t,a} + s_a] > r_c \tag{2}$$

where $r_{t,a}$ is now the true financial rate of return to the developmental use of the land, and s_a is the subsidy applied to it. The chances that $r_{a,t}$ is actually greater than r_c is now reduced: subsidies are distorting the comparison. Unless the subsidies themselves have powerful justifications, the subsidies are effectively stimulating economically unwarranted land conversion and biodiversity loss. They may be acting as perverse incentives (OECD, 1999).

A second issue is the one addressed in this handbook: r_c comprises two elements, the market rate of return to conservation and the non-market benefits, i.e.

$$r_c = r_{c,m} + r_{c,nm} \tag{3}$$

If only $r_{c,m}$ show up in the market place, then the comparison of developmental and conservation rates of return will tend to favour the developmental use. There are therefore two stages to the correction of this problem: (a) placing a value on $r_{c,nm}$ and (b) possibly converting that value into a resource flow through the creation of markets (OECD, forthcoming).

Third, the proper context for the comparison of rates of return is one of uncertainty. There will be considerable certainty about the benefits of developmental uses of land: we are familiar with the yields of crops, the harvest of timber, number of houses per hectare and their relevant prices. As all writers on biodiversity stress, however, we are not confident that we know the benefits of biodiversity conservation and sustainable use. The context is one of uncertainty rather than risk. Moreover, the uncertainty has some asymmetry about it - diversity in itself is unlikely to be bad - but the policy of conserving diversity might well have a cost in terms of foregone productivity, as earlier chapters suggested. Most of that opportunity cost of conservation, however, appears in the rate of return to developmental uses of the land. At the very least then there is an option value that needs to be added to the conservation side of the inequality. Hence the returns to conservation become:

$$r_c = r_{c,m} + r_{c,nm} + r_{c,ov} \quad (4)$$

The economic approach therefore requires that a careful evaluation of conservation benefits include all non-market benefits, inclusive of the value of diversity itself as an option. Expressed this way, the economic approach offers a potentially powerful means of conserving more rather than less biodiversity. Obviously, the same effect could be achieved by some from of command-and-control policy that was not informed by economic benefit measurement, but, as noted earlier, such approaches have far less potential for determining the 'optimal' degree of diversity protection.

Much the same analysis applies to the issue of sustainable use of biological resources. The Convention on Biological Diversity acknowledges that sustainable use may often be the most efficient route to conservation. The essence of the argument is that outright protection may be costly and difficult to enforce. Sustainable use on the other hand involves persuading existing users of biodiversity to manage the resource sustainably rather than unsustainably. Sometimes, information and technology transfer will achieve this transition, but in many cases the problem derives from the fact that the short-term gains from unsustainable use are higher than the short-term gains from sustainable use. Even though the long-term gains from sustainable use exceed those of unsustainable use, the short-term dominates. This arises from many features of land use, for example insecure property rights. Insecurity discourages long-term investment in the land since the 'owner' can be dispossessed at any time. Such context are also typified by very high discount rates since land users are compelled to worry about net benefits from land use in the immediate future and not in the far future.

Economic approaches to valuation are relevant here since they can determine the compensation that land users would need to switch from unsustainable to sustainable use. This compensation will equal the opportunity cost of sustainable use and management. Various analyses have demonstrated these procedures for estimating compensation in the context of slash and burn agriculture (Schneider, 1995) and sustainable forestry (Pearce *et al.*, 2001).

10.3 Precautionary approaches

All decision-making about biodiversity needs to be sensitive to the issue of uncertainty about the value of biodiversity and one of the most effective ways to diminish uncertainty is to gather adequate information. Methodologies which may be common to non-instrumental and instrumental approaches include the notion of a safe minimum standard (SMS) and the precautionary principle (PP). Neither approach produces a notion of the quantitative scale of conservation that is justified, but both invert the usual notion that conservation and loss of biodiversity have equal status, or that ‘development’ is superior to conservation. Both argue that the presumption of policy should be that biodiversity should be conserved. In the case of the SMS approach, this presumption should be relaxed if and only if the opportunity cost of conservation is, in some sense, very large. The latter requires that, since we know so little about the importance of biodiversity, this uncertainty should dictate a very cautious attitude towards its destruction.

Safe minimum standards

Strictly, SMS refers to the minimum level of preservation that ensures survival (Ciriacy-Wantrup, 1968; Bishop, 1978). While the full value of a species or an ecosystem function may not itself be measurable, it is known to be positive on the grounds that species or functions previously thought to be ‘useless’ have proved to be ‘useful’. Hence something that has positive value should be sacrificed only if the benefits of that sacrifice are considerable. The burden of proof thus falls on those who wish to destroy biodiversity to demonstrate that the sacrifice is worthwhile. As Randall (1986) notes, the approach tends to redefine the problem rather than provide an answer. The presumption that all species are valuable fits some of the non-instrumental views of value, but sits uneasily with instrumental views and perhaps even with ecologists’ notions of redundant species. Most telling is the problem that, if species value is not measured, it is hard to decide whether the cost of conservation is outweighed by its benefit. In short, there is no guidance on what a ‘very high’ opportunity cost of conservation is. Nonetheless, the SMS approach is instructive in forcing attention on the benefits of the processes that give rise to biodiversity loss. In a very large number of cases it seems clear that land conversion confers extremely low benefits on those making the conversion, and many costs in addition to biodiversity loss. This suggests that considerable amounts of diversity could be conserved at low cost through measures to compensate those who convert land, or to provide them with alternative livelihoods. Other contexts are far more problematic, as with, for example, the cost associated with peri-urban green area conservation. This cost shows up most readily in the price of land for, say, residential purposes.

The precautionary principle

The precautionary principle, like the SMS approach, offers a different perspective but not one that deals with the scale of conservation. The principle itself is defined very differently in different contexts. In its strictest interpretation it suggests that no action should be taken if there is any likelihood at all, however small, that significant biodiversity loss could occur. This likelihood may be independent of the scientific evidence. That is, unless there is certainty that there are no losses, actions should not be taken which, for example, release harmful pollutants into the environment. Perhaps the closest form of the strict PP in practice is the German *Vorsorgeprinzip* - widely translated as the precautionary principle - which is designed to secure *Umweltschutz,* environmental protection. *Umweltschutz* is a constitutional obligation in some German states, but not a Federal obligation. *Vorsorge* developed as a justification for state intervention as part of the social democratic movement and as a counter to the prevailing 1970s philosophy that limited environmental protection on cost grounds. *Vorsorge* requires that environmental risks be detected early (the research focus), that action be taken even without proof of damage when irreversibility is feared, that technology should be developed for preventive action, and that the state has the obligation of environmental protection. There appears to be no mention of cost in this interpretation of the *Vorsorgeprinzip*.

Construed in this way, the precautionary principle can be thought of as one approach to the ‘zero-infinity’ problem in which the probability of damage is small or unknown, but the consequences are potentially very large (Page, 1978; Camerer and Kunreuther, 1989). As such, the precautionary principle can be held to apply to both risk and uncertainty contexts, the former being one where probabilities are known, the latter where they are not known.

A second interpretation of the PP requires that there be a presumption in favour of not harming the environment unless the opportunity costs of that action are very high, i.e. the safe minimum standards rule identified above.

Yet further interpretations of the PP suggest that it applies particularly where there are good grounds for judging either that action taken promptly at comparatively low cost may avoid more costly damage later, or that irreversible effects may follow if damage is delayed. Rather than focusing on the need for ‘high’ benefits to justify the degrading activity, this last definition emphasises that the cost of the protective measure should be low relative to the expected environmental gain.

On some of the interpretations, adoption of the precautionary principle could be expensive. If the benefits forgone are substantial and new information reveals that the measure turns out not to have been warranted, then there will be a high net cost to precaution. On the other hand, if new information reveals that precaution was justified, nothing is lost. This suggests that some balancing of costs and benefits still must play a role even in contexts where the precautionary principle is thought to apply.

Probably the most important feature of the SMS and PP approaches is their presumption that biodiversity has 'high' value, regardless of whether that value is formulated in an instrumental or non-instrumental fashion. Both are also grounded in the context of uncertainty (rather than risk) which in itself is sufficient to justify cautious approaches, especially if ecologists are right in warning that loss of interconnectedness between species and loss of ecosystem resilience could be potentially very damaging.

10.4 Setting priorities for biodiversity conservation revisited: cost-benefit analysis

Chapter III illustrated a cost-effectiveness procedure for determining priorities for biodiversity conservation**.** An alternative procedure for setting priorities discussed in Chapter VI uses cost-benefit analysis. The procedure would be:

a. assess the money value of diversity conservation;

b. assess the costs of a conservation policy;

c. compare benefits and costs such that policy interventions would be ranked according to their benefit-cost ratios (B/C);

d. all policies with B/C ratios greater than unity would be potentially judged worthwhile;

e. the first ranked policy would be undertaken, followed by the second, and so on until the 'conservation budget' is exhausted. To complement the ranking rate-of-return on investment where feasible can be used.

The money value of biodiversity conservation would comprise the use and non-use values indicated in earlier chapters. Included in use values would be all the values directly related to use, such as genetic information for pharmaceuticals and seed varieties, and the indirect values of ecosystems as providers of services and the role of diversity in fostering resilience to shocks and stresses. Included in non-use values would be the willingness to pay of individuals for biological resources and diversity independently of any uses made of those resources.

The cost-benefit approach assumes that at least a significant part of the functions and services provided by diversity can be measured in economic terms. For this to be the case individuals must have identifiable preferences for (or against) those functions and services, and one or more of the methodologies set out in Chapters VII and VIII must be applicable.

Randall (1991) considers some of the criticisms of the cost-benefit approach:

a. the cost-benefit approach is founded in the instrumentalist view of value, whereas the ‘true’ notion of value is intrinsic. Instrumental values are variable - preferences might change, dictating loss of diversity, whereas intrinsic values are constant through time;

b. technological change may enable the uses of biodiversity to be met by some other means - bio-technology would be a case in point. This might reduce the ‘demand’ for biodiversity and hence put greater pressure on it (Ehrenfeld, 1988). A non-instrumental view would confer value independent of the state of technology;

c. cost-benefit is an incremental procedure: it values small or discrete changes in the stock of biodiversity, whereas the total stock has extremely high value. Cost-benefit might be consistent with judging each small loss of biodiversity as being justified, but each small change contributes to the risk that the total stock will be lost (Norton, 1988);

d. cost-benefit embodies the economist’s notion that value is relative, i.e. the value of something is always relative to something else. Critics argue that biodiversity has absolute value in itself. Its value cannot be measured relative to other things.

Criticism (a) reflects the different philosophical viewpoints on value as discussed in Chapter III. Even if non-instrumental values are thought to be constant through time, it was noted above that non-instrumental views are difficult to translate into practical policy in the context of the threats to biodiversity and the opportunity cost of resources used to conserve biodiversity. But concepts of intrinsic value may also change through time.

Criticism (b) has some foundation, but there are views that suggest technological change will benefit biodiversity. In particular, genetic modification of crops reduces the chances of failures in production and also promises higher yields. In the context of a growing demand for agricultural products, higher yields would reduce the demand for further land to accommodate more crops. Since land conversion is the main cause of biodiversity loss, the effect would be to conserve biodiversity. Much therefore depends on the balance between this potentially beneficial effect and potentially harmful effects from biotechnology via species interaction and cross-pollination.

The view that incremental approaches to value could result in total losses has some validity. The argument against it is that as the stock gradually diminishes so the theory of economic valuation would dictate that the value of each remaining element is increased, i.e. increasing scarcity confers increasing value. This would hold so long as there is always some decision-maker taking a holistic view, or if markets in biodiversity function properly. The absence of markets and the potential for irrationality in decision-making makes the chances of total damage a realistic possibility. Additionally, those ecologists who stress the potential discontinuities in ecological systems would argue that small changes may produce large effects if the change occurs in the region of an unstable ecosystem equilibrium. It is important to remember that economic valuation relates to small and discrete changes.

The final criticism reflects another feature of some of the non-instrumental approaches to biodiversity. Since biodiversity is ‘the web of life’, the idea that it can be traded against other things strikes many as a logical error. Effectively, conservation of biodiversity becomes a categorical imperative, a pre-eminent moral rule. Randall (1991) notes that the case for such pre-eminence has not been made, particularly in the necessary context of opportunity cost.

Clearly, cost-benefit analysis is not an uncontroversial procedure, but then neither is any of the alternatives.

It may be possible to mix policy approaches, i.e. to combine instrumental procedures such as cost-benefit analysis and the non-instrumental procedures that involve some notion of intrinsic value. Several writers have argued this, e.g. Page (1977), Pearce (1976) and Randall (1991). The basic idea would be to adopt a broad rule that ensures sustainability or ‘permanent liveability’ as Page calls it. Such rules would set bounds on the use of natural resources, including biodiversity, in the name of sustainability. Within those bounds, cost-benefit approaches would be appropriate. In Randall’s terminology, the SMS would become the bound, and cost-benefit would operate subject to SMS limitations.

The practical issue is what such mixed rules would be like in terms of decision rules. Since the SMS principle is not formulated in quantitative terms but the cost-benefit rule is, there are obvious problems of combining the two rules. It is possible that the rules could be mixed by adopting a modified cost-benefit rule, one that effectively adopts cost-benefit guidelines but subject to the rule that the sum total of all projects and policies must not leave the total environment (or biodiversity, whichever is the focus) degraded. Rules of this kind - so-called ‘strong sustainability’ rules - have been formulated in theory (Barbier *et al.*, 1990; Pires, 1998) but it is less easy to see how they would work in practice. There is a substantial research agenda that could be aimed at mixing the various approaches to conserving biodiversity (OECD, 2001 c).

10.5 Focusing conservation policies: species or ecosystems?

While much of the discussion about biodiversity policy is couched in terms of species conservation, it is clear that none of the approaches discussed so far warrants an exclusive focus on individual species. The economic approach might be justified in these terms because individuals often do focus on single species issues. Increasingly, however, there is a recognition that it is whole systems that need to be conserved, even where there might be a case for supposing that a single species has high charismatic or symbolic value. The system-wide focus is reinforced by the ecological value approaches since, as Chapter II noted, they regard species-interaction as more important than individual species. Similarly, the approaches based on a ‘diversity function’ are consistent with individual species being neglected so long as genetic diversity is conserved or some form of species representativeness is preserved. This is well recognised by the CBD process, which undertook the ecosystem approach as one of its guiding strategies. As a strategy for the integrated management of land, water and living resources, the ecosystem approach helps to reach a balance of the three convention objectives. In effect, the CBD process also recognises the relationship between the ecosystem approach and valuation. COP Decision V/6, which elaborates on the ecosystem approach to include principles and operational guidance, underscores the importance of proper valuation of ecosystems goods and services for the ecosystem approach. In essence, it highlights the topic of this manual.

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