THE ECONOMIC VALUE OF BIODIVERSITY
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IN ASSOCIATION WITH THE BIODIVERSITY PROGRAMME OF IUCN — THE WORLD CONSERVATION UNION

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The World Conservation Union

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PREFACE

This book began life as a report to the International Union for the Conservation of Nature (IUCN) (the World Conservation Union) in Gland, Switzerland, in 1993. IUCN is internationally renowned for its pioneering work on nature conservation. Jeff McNeely, IUCN's Chief Conservation Officer, has long shown an appreciation of the role that economics has to play in habitat and species conservation (see, for example, his own *Economics and Biological Diversity,* IUCN, Gland, 1988). Our original remit was to look just at the issue of *economic value,* ie the kinds of economic values that are generated by conservation activity but which may well not be captured in the market place. The result of this `failure' to capture such economic values is a distortion, a tilted playing field with the odds stacked against conservation and in favour of the economic activities that destroy biological resources. In this book version of our report we have gone further and have asked *why biodiversity disappears* and how its economic value might be captured by various institutional mechanisms. The theme of the volume is therefore roughly as follows:

- economic forces drive much of the extinction of the world's biological resources and biological diversity; yet
- biodiversity has economic value. If the world's economies are rationally organized, this suggests that biodiversity must have less economic value than the economic activities giving rise to its loss;
- yet we know that many biological resources do have significant economic value. We also know that many of the destructive activities themselves have very low economic value; therefore
- something is wrong with the way actual economic decisions are made — for some reason they fail to `capture' the economic values that can be identified;
- these `economic failures' lie at the heart of any explanation for the loss of biological diversity. If we can address them, there is a chance of reducing biodiversity loss.

Our perspective is therefore peculiarly economic. We make no apologies for that. We do not imply that other disciplines are not relevant or not important. We focus on the economics because it is our area of expertise and because we believe, very strongly, that the `economic paradigm' illuminates the debate over biodiversity loss.

Those familiar with our earlier books will detect the common theme. There is a strong and pervasive set of links between economy and environment. Failure to understand those links, complex though they often are, is a failure to understand the primary motive forces for environmental destruction. That failure carries through to the design of policies which have little real chance of making significant impacts on the loss of the world's environments. These themes are spelled out in detail in David Pearce, Anil Markandya and Edward Barbier, *Blueprint for a Green Economy* (1989); David Pearce, Edward Barbier and Anil Markandya, *Sustainable Development: Economics and the Environment in the Third World* (1990); David Pearce (editor), *Blueprint 2: Greening the World Economy* (1991); David Pearce (editor), *Blueprint 3: Measuring Sustainable Development,* (1993); David Pearce, *Economic Values and the Natural World* (1993), all published by Earthscan; and David Pearce and Jeremy Warford, *World Without End: Economics, Environment and Sustainable Development* (Oxford University Press, 1992).

Those seeking a guide to environmental economics can consult R Kerry Turner, David Pearce and Ian Bateman, *Environmental Economics: an Elementary Introduction* (Harvester Wheatsheaf, Hemel Hempstead, 1994).

David Pearce and Dominic Moran

London, June 1994

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Special thanks to David Pearce's own stock of feline biological diversity — Blueprint, Floss and Dill — who continue to entertain.

1 INTRODUCTION

The loss of the world's biological diversity is causing major concern worldwide. That concern is embodied, for example, in the Convention on Biological Diversity negotiated at Rio de Janeiro in 1992. But how important is biodiversity? And how can priorities for conservation be determined? This volume is concerned with one important approach to these questions: the economic approach. The central argument is that by ascribing economic value to biodiversity, more powerful, more practical arguments can be formulated for its conservation. The book reviews the economic approach and the available evidence on the economic value of biodiversity

The term `biological diversity', often shortened to `biodiversity', is an umbrella term used to describe the number, variety and variability of living organisms in a given assemblage. Biodiversity therefore embraces the whole of `Life on Earth'. Decline in biodiversity includes all those changes that have to do with reducing or simplifying biological heterogeneity, from individuals to regions. This is a more subtle definition than the global stock of *biological resources,* a more anthropocentric term for biota such as forests, wetlands and marine habitats. Biological resources are simply those components of biodiversity which maintain current or potential human uses. They represent the diversity about which most is known. This anthropocentric view of biological resources offers a convenient `window' for economic analysis over alternative value paradigms such as `intrinsic value': values in themselves and, nominally anyway, unrelated to human use. Intrinsic values are relevant to conservation decisions, but they are generally not measurable. As such they do not help to define actions in the context where choices have to be made against the backdrop of scarce conservation funds.

This chapter explains some of the key concepts of biodiversity and approaches to the measurement of biodiversity and its components. Some tentative estimates of rates of extinction are presented and qualified by a discussion of the scientific uncertainties which complicate these estimates and the choice of reliable indicators. Institutional and economic forces driving depletion are outlined prior to an introduction of the value categories relevant to biological resources. The chapter stresses the range of measures of diversity from different scientific perspectives. The different conceptualizations of biodiversity lead to different policy prescriptions, and require different indicators for monitoring and assessment.

THE MEANING OF BIOLOGICAL DIVERSITY

Biodiversity may be described in terms of genes, species, and ecosystems, corresponding to three fundamental and hierarchically-related levels of biological organization.

Genetic diversity

Genetic diversity is the sum of genetic information contained in the genes of individuals of plants, animals and micro-organisms. Each species is the repository of an immense amount of genetic information. The number of genes range from about 1000 in bacteria, up to 400 000 or more in many flowering plants. Each species is made up of many organisms, and virtually no two members of the same species are genetically identical. This means for example that even if an endangered species is saved from extinction, it will probably have lost much of its internal diversity. When the populations are allowed to expand again, they will be more genetically uniform than their ancestral populations. For example, the bison herds of today are biologically not the same in terms of their genetic diversity as the bison herds of the early 18th century (McClenagham et al, 1990).

Population geneticists have developed mathematical formulae to express a genetically effective population size. These explain the genetic effects on populations which have passed through a `bottleneck' of a small population size, such as the North American bison or African cheetah (WCMC, 1992). The resultant inbreeding may have a number of detrimental effects such as lowered fertility and increased susceptibility to disease. This is termed `inbreeding depression'. The effects of small population size depend on the breeding system of the species and the duration of the bottleneck. If the bottleneck lasts for many generations, or population recovery is very slow, a great deal of variation can be lost. The converse, `outbreeding depression', occurs when species become genetically differentiated across their range, and then individuals from different parts of the range breed.

Genetic differentiation within species occurs as a result of either sexual reproduction, in which genetic differences from individuals may be combined in their offspring to produce new combinations of genes, or from mutations which cause changes in the DNA.

The significance of genetic diversity is often highlighted with reference to global agriculture and food security. This stresses the reliance of the majority of the world's human population on a small number of staple food species, which in turn rely on supply of genes from their wild relatives to supply new characteristics, for example to improve resistance to pests and diseases (Cooper et al, 1992).

Species diversity

Species are regarded as populations within which gene flow occurs under natural conditions. Within a species, all normal individuals are capable of breeding with the other individuals of the opposite sex belonging to the same species, or at least they are capable of being genetically linked with them through chains of other breeding individuals. By definition, members of one species do not breed freely with members of other species. Although this definition works well for many animal and plant species, it is more difficult to delineate species in populations where hybridization, or self-fertilization or parthenogenesis occur. Arbitrary divisions must be made, and indeed this is an area where scientists often disagree.

New species may be established through the process of *polyploidy,* the multiplication of the number of gene-bearing chromosomes, or more commonly, as a result of geographic speciation. This is the process by which isolated populations diverge by evolution as a result of being subjected to different environmental conditions. Over a long period of time, differences between populations may become great enough to reduce interbreeding and eventually populations may be able to co-exist as newly formed, separate species. Within the hierarchical system used by scientists to classify organisms, species represent the lowest rung on this ladder of classification. In descending order, the main categories, or taxa, of living things are:

Kingdom

Phylum

Class

Order

Family

Genus

Species

We do not know the true number of species on earth, *even to the nearest order of magnitude.* Wilson (1988) estimates that the absolute number of species falls between 5 and 30 million, although some scientists have put forward even higher estimates, up to 50 million. At present approximately 1.4 million living species of all kinds of organisms have been described. The best catalogued groups include vertebrates and flowering plants, with other groups relatively under-researched, such as lichens, bacteria, fungi and roundworms. Likewise, some habitats are better researched than others, and coral reefs, deep ocean floor and tropical soils are not well studied. This lack of knowledge has considerable implications for the economics of biodiversity conservation, particularly in defining priorities for cost-effective conservation interventions.

The single most obvious pattern in the global distribution of species is that overall species richness increases with decreasing latitude. Not only does this apply as a general rule, it also holds within the great majority of higher taxa, at order level or higher. However, this overall pattern masks a large number of minor trends. Species richness in particular taxonomic groups, or in particular habitats, may show no significant latitudinal variation, or may actually decrease with decreasing latitudes. In addition, in terrestrial ecosystems, diversity generally decreases with increasing altitude. This phenomenon is most apparent at extremes of altitude, with the highest regions at all latitudes having very low species diversity (although these areas also tend to be of limited size, which may be one factor resulting in lower species numbers). In terms of marine systems, depth is the analogue of altitude in terrestrial systems and biodiversity tends to be negatively correlated with depth. Gradients and changes in species richness are also noticeably correlated to precipitation, nutrient levels and salinity, as well as other climatic variations and available energy.

Ecosystem diversity

Ecosystem diversity relates to the variety of habitats, biotic communities and ecological processes in the biosphere as well as the diversity within ecosystems. Diversity can be described at a number of different levels and scales:

- Functional diversity is the relative abundance of functionally different kinds of organisms.
- Community diversity is the number sizes and spatial distribution of communities, and is sometimes referred to as patchiness.
- Landscape diversity is the diversity of scales of patchiness.

No simple relationship exists between the diversity of an ecosystem and ecological processes such as productivity, hydrology, and soil generation. Neither does diversity correlate neatly with ecosystem stability, its resistance to disturbance and its speed of recovery. There is no simple relationship within any ecosystem between a change in its diversity and the resulting change in the system's processes. For example, the loss of a species from a particular area or region (local extinction or extirpation) may have little or no effect on net primary productivity if competitors take its place in the community. The converse may be true in other cases. For example, if herbivores such as zebra and wildebeest are removed from the African savanna, net primary productivity of the ecosystem decreases.

Despite these anomalies, Reid and Miller (1989) suggest six general rules of ecosystem dynamics which link environmental changes, biodiversity and ecosystem processes.

1. The mix of species making up communities and ecosystems changes continually.
2. Species diversity increases as environmental heterogeneity or the patchiness of a habitat does, but increasing patchiness does not necessarily result in increased species richness.
3. Habitat patchiness influences not only the composition of species in an ecosystem, but also the interactions among species.
4. Periodic disturbances play an important role in creating the patchy environments that foster high species richness. They help to keep an array of habitat patches in various successional states.
5. Both size and isolation of habitat patches can influence species richness, as can the extent of the transition zones between habitats. These transitional zones, or `ecotones', support species which would not occur in continuous habitats. In temperate zones, ecotones are often more species rich than continuous habitats, although the reverse may be true in tropical forests.

6 Certain species have disproportionate influences on the characteristics of an ecosystem. These include *keystone species,* whose loss would transform or undermine the ecological processes or fundamentally change the species composition of the community.

The discussion has shown how biodiversity is a very complex and all-embracing concept, which can be interpreted and analysed on a number of levels and scales. The next section examines some approaches to measuring these concepts.

MEASUREMENT OF BIODIVERSITY

A better understanding of biodiversity can be obtained when we examine exactly what we measure in order to assess biological diversity. However, this also serves to highlight further the range of interpretations, and the importance placed on different hierarchical levels of biodiversity by scholars of different disciplines, and by policy makers. Reid *et al* (1992) have commented that there is even now no clear consensus about how biodiversity should be measured. Indeed, debates on the measurement of biodiversity have filled a substantial part of the ecological literature since the 1950s. This lack of consensus also has important implications for the economics of biodiversity conservation. At its most basic level, any measure of cost-effectiveness used to guide investments in conservation must have some index or set of indices of biodiversity change. In the following sections, some aspects of measurement of biodiversity are examined, distinguishing the same components of biodiversity: genetic diversity, species diversity and ecosystem diversity.

Measurement of genetic diversity

The analysis and conceptualization of differences within and among populations is in principle identical regardless of whether we are considering a `population' to be a local collection of individuals, geographical race, subspecies, species, or higher taxonomic group. Genetic differences can be measured in terms of phenotypic traits, allelic frequencies or DNA sequences.

Phenetic diversity

Phenetic diversity is based on measures of phenotypes, individuals which share the same characteristics. This method avoids examination of the underlying allelic structure. It is usually concerned with measurement of the variance of a particular trait, and often involves readily measurable morphological and physiological characteristics. Phenetic traits can be easily measured, and their ecological or practical utility is either obvious or can be readily inferred. However, their genetic basis is often difficult to assess, and standardized comparisons are difficult when populations or taxa are measured for qualitatively different traits.

Allelic diversity

The same gene can exist in a number of variants and these variants are called alleles. Measures of allelic diversity require knowledge of the allelic composition at individual loci. This information is generally obtained using protein electrophoresis, which analyses the migration of enzymes under the influence of electric field. Allelic diversity may be measured at the individual level, or at the population level. In general, the more alleles, the more equitable their frequencies, and the more loci that are polymorphic, the greater the genetic diversity. Average expected heterozygosity (the probability that two alleles sampled at random will be different) is commonly used as an overall measure. A number of different indices and coefficients can be applied to the measurements to assess genetic distance (see Antonovic, 1990). The detection of allelic variation by electrophoresis has the advantage that it can be precisely quantified to provide comparative measures of genetic variation. However, the disadvantages are that it may not be representative of variation in the genome as a whole, and does not take account of functional significance or selective importance of particular alleles.

Sequence variation

A portion of DNA is sequenced using the polymerase chain reaction technique (PCR). This technique means that only a very small amount of material, perhaps one cell, is required to obtain the DNA sequence data, so that only a drop of blood or single hair is required as a sample. Closely related species may share 95 per cent or more of their nuclear DNA sequences, implying a great similarity in the overall genetic information.

Measurement of species diversity

Species diversity is a function of the distribution and abundance of species. Often, species richness — the number of species within a region or given area — is used almost synonymously with species diversity. However, technically, species diversity includes some consideration of evenness of species abundances. Let us first consider species richness as a proxy measure of species diversity.

In its ideal form, species richness would consist of a complete catalogue of all species occurring in the area under consideration, but this is not usually possible unless it is a very small area. Species richness measures in practice, therefore, tend to be based on samples. Such samples consist of a complete catalogue of all organisms within a taxa found in a particular area, or of a measure of species density in a given sample plot, or a numerical species richness defined as the number of species per specified number of individuals or biomass.

A more informative measure of diversity would also incorporate the `relatedness' of the species in a fauna (Williams et al, 1991; Reid et al, 1992). Using a measure of species richness would imply that a region containing many closely related species would be preferred to one containing a fractionally smaller number of distantly related or genealogically unrelated species. Alternative measures being developed augment species richness with measures of the degree of genealogical difference. Derived from cladistic methods, these measures include the weighting of close-to-root species, higher-taxon richness, spanning-tree length and taxonomic dispersion (Williams et al, 1991). Close-to-root species and higher-taxon richness explicitly use polarity from the root of the cladogram to weight higher-ranking taxa or `relic' species as distinct survivors of long-independent lineages and original conduits of genetic information. In contrast, spanning tree length and taxonomic dispersion are more general tree measures of sub-tree `representativeness'. Polarity from the root of the tree is less important than the amount of the cladogram represented by a fauna or the choice of a fauna to evenly cover the diversity of subgroups found in the cladogram. There is considerable disagreement as to which measure best characterizes the pattern of difference in the popular concept of biodiversity, although there is considerable support for taxonomic dispersion as a method of selecting faunas which most evenly represent a variety of cladogram sub-groups. For the time being, difficulties in actual implementation of cladistic measures suggest reliance on cruder indicators of richness of genera or families for rapid assessment of species diversity.

Measurement of community diversity

Many environmentalists and ecologists put emphasis on conservation of biodiversity at the community level. There are a number of factors which make measurement and assessment of diversity at this level more nebulous and less clearly defined. Many different `units' of diversity are involved at the supra-species level, including the pattern of habitats in the community, relative abundance of species, age structure of populations, patterns of communities on the landscape, trophic structure, and patch dynamics. At these levels, unambiguous boundaries delineating units of biodiversity do not exist. By conserving biodiversity at the ecosystem level, not only are the constituent species preserved, but also the ecosystem functions and services protected. These include pollutant cycling, nutrient cycling, climate control, as well as non-consumptive recreation, scientific and aesthetic values (see for example, Norton and Ulanowicz, 1992).

Given the complexities of defining biodiversity at community or ecosystem level already described, there is a range of different approaches to measuring ecosystem diversity. As Reid et al (1992) explain, any number of community attributes are components of biodiversity and may deserve monitoring for specific objectives. There are several generic measures of community level diversity. These include biogeographical realms or provinces, based on the distribution of species, and ecoregions or ecozones, based on physical attributes such as soils and climate. These definitions may differ according to scale. For example, the world has been divided into biogeographical provinces, or more fine-grained classifications which may be more useful for policy-making. More policy orientated measures include the definition of `hotspots', based on the number of endemic species, and `megadiversity' states.

These concepts will be discussed in the context of using indicators for assessing and monitoring biodiversity. The following section introduces extinction, and some of the estimates of current rates of species extinction which have resulted in urgent need for conservation of global biodiversity.

THE RATE OF BIODIVERSITY LOSS

Estimates of precise rates of loss of biological diversity are hampered by the absence of any baseline measurement. However, from evidence of island habitats it seems likely that the expansion of the human niche by various forms of conversion is geometrically related to extinctions. Further recent evidence from observation of potential `indicator' species such as amphibians and birds provides some indication of accelerated loss in excess of historical or background rates (Pechmann et al, 1991; Myers 1993). Table 1.1 shows some estimates of current rates of species extinction based on extrapolations of human land use trends related to species area curves which are the basis of island biogeography. Over the next century the projected loss of species might be expected to be as high as 20 to 50 per cent of the world's totals which represents a rate between 1000 to 10000 times the historical rate of extinction (Wilson, 1988). The rate of loss is outstripping the natural regenerative capacity of evolution to throw up new or evolved species. The extinction `outputs' far exceed the speciation `inputs' (Ehrlich and Ehrlich, 1992). The potential effects of accelerated extinction and depletion of the genetic base may be discerned over varying time horizons. In the long term, processes of natural selection and evolution may be dependent on a diminished resource base, simply because fewer species are being born. The implications of species depletion for the integrity of many vital ecosystems are far from clear. The possible existence of depletion thresholds, associated system collapse, and huge discontinuities in related social cost functions, are potentially the worst outcome in any reasonable human time horizon. Such scenarios are indicative of the links between ecosystem integrity and economic well-being. More immediately, the impoverishment of biological resources in many countries might also be regarded as an antecedent to a decline in community or cultural diversity, indices of which are provided in diet, medicine, language and social structure (Harmon, 1992).

Table 1.1 *Estimates of the current rates of species extinction*

Estimate of loss of species (%)	Basis	Source
33–50 by year 2000	Forest area loss	Lovejoy (1980)
50 by year 2000	Forest area loss	Ehrlich (1981)
25–30 in 21st century	Forest area loss	Myers (1989)
33 in 21 st century	Forest area loss	Simberloff (1986)

At least four questions emerge from the scientific uncertainty surrounding species loss.

1 What is the number of species from which to measure current rates of loss and the detection of this rate, allowing for background evolutionary turn-over?

2 How sound are the principals and predictions of island biogeography and by how much are current extinction estimates (probably) understated?

3 Given the likely time horizons at issue, can we be concerned with the perversion of the evolutionary processes as opposed to the immediacy of system thresholds and flips?

4 What is the potential for using indicator species or a more sophisticated index to guide conservation efforts, and is there any scientific consensus on appropriate species or ecosystems to be used?

The need to pursue cost-effective investment interventions in biodiversity conservation has added considerable urgency to these issues, and the indicators debate in particular. Reliance on pivotal key-stone or umbrella species (Noss et al, 1992) is appealing but crude. Similarly focusing on wider taxonomic groups or ecosystem functions provides few indications of the likelihood of successful interventions, given wider socio-economic pressures on wilderness ecosystems and protected areas. Criteria such as species sensitivity to habitat disruption or poor reproductive capacity can be combined with other socio-economic data such as population density, deforestation or figures on conservation investment expenditure, to provide some indication of where species are threatened. However, assuming some consensus definition of threat raises the issue of whether funding is most effectively directed to those areas most under threat, or away from them entirely in favour of areas with a higher likelihood of success. This in turn implies some objective assessment of a `successful' intervention. Given that no species can be saved indefinitely, the objective decision criterion becomes

the extra cost of an increment to the probability of survival (Montgomery et al 1994; Hohl and Tisdell, 1993).

Understandably the development of investment criteria designed to maximize diversity per dollar and incorporating a composite threat indicator is likely to take time. Inevitable data restrictions are certain to further complicate what is already a contentious exercise. Nevertheless, two facts are clear. At some point a consensus measurement of biodiversity is required to guide the investment of scarce funds. The resulting index may seem arbitrary and will inevitably contravene some section of scientific opinion but will be necessary to provide a general direction for biodiversity investment. Moreover, using this index, any cost-effective system of area triage will necessarily require some consideration of complementarity of resulting faunal designations. In other words, the selection of successive areas for protection, ideally needs to be based on the incremental complement to diversity afforded by the last fauna until the complement is reduced to zero (Vane-Wright et al, 1991; Faith, 1994). This is clearly a massive undertaking, requiring precise taxonomic inventories and as much socio-economic information as dictated by the guiding index. At the same time such a process could show how an excessive concentration on certain biota can yield diminishing returns. The process of building on rapid appraisal rules of thumb such as hot spot or mega-diverse areas has already begun (see Dinerstein and Wikramanayake, 1993; Pearce et al, 1994). Emerging prescriptions are considerably less discriminating than the precise genealogical indices outlined in this chapter, and do not as yet attempt any faunal complementarity ranking. They do, nevertheless, attempt to combine basic species richness indicators with the socio-economic parameters most immediate to biodiversity loss.

STRUCTURE OF THE BOOK

The following chapters take this largely scientific review as a point of departure for understanding the loss of biological diversity. Chapter 2 introduces core themes of fundamental causation and economic value and explains how these help to clarify some of the policy issues relevant to the biodiversity conservation debate. Chapter 3 sets the issue in the context of cost-benefit analysis. Chapter 4 looks at the causes of biodiversity loss. Chapter 5 summarizes the methodologies for economically valuing environmental resources. Chapter 6 summarizes the empirical literature on those values. Chapter 7 investigates the other side of the conservation issue — the economic value of the development alternative. Chapter 8 explores the issue of emerging global markets for biological resource conservation. Chapter 9 draws the various issues together.

2 SAVING BIODIVERSITY: AN OVERVIEW OF CAUSAL FACTORS

In summer 1992 in Rio de Janeiro, the world's nations agreed a global Convention on Biological Diversity. It aims to protect the world's biological resources from further erosion or, at least, to slow that rate of erosion down. Yet the rate of erosion of biodiversity is increasing. Despite the Convention, the need to demonstrate the importance of conservation remains as strong as it ever was, perhaps stronger. One aspect of the process of changing government and popular perceptions about biological resources is to show that the sustainable use of biodiversity has positive economic value, and that this economic value will often be higher than the value of alternative resource uses which threaten biodiversity. Subsequent chapters are concerned with three fundamental characteristics of the biodiversity debate:

1 *demonstrating the economic values of biological resources* in the contexts where the values are often not reflected in market processes;

2 explaining why, despite those economic values, *biodiversity continues to be threatened;*

3 finding ways to *capture or realize economic value.*

Recognition of a broader total economic valuation of natural assets can be instrumental in altering decisions about their use, particularly in investment and land-use decisions which present a clear choice between destruction or conservation. Such decisions are being faced in both developed and developing countries, where a host of competing social and economic claims increasingly conflict with the resource demands of area protection. Greater understanding of the functioning of natural ecosystems combined with enhanced valuation techniques are an increasing influence on national conservation strategies, while international and multilateral initiatives emphasize the global dimension to the issue of biodiversity loss. We argue that addressing the economic causes of biodiversity loss is

extremely important if the world really does want to slow down the erosion of its biological resources. Much of the biodiversity that needs saving resides in the developing world. Since biodiversity conservation is not, understandably, a priority for the developing world, the resources needed for conservation must come from the North, while the political commitment must come from the South and North alike. However we would like the world to be, the brute fact is that only policies which offer mutual self-interested gains to both North and South stand a chance of succeeding. In the longer term we may hope for changes of attitudes and priorities in the world generally, especially as incomes rise in the South. But relying on such changes to bring about conservation is foolhardy and counterproductive. That is why the economic approach matters. It does emphasize mutual economic gain as the foundation for the solution to the biodiversity problem.

The main reason for the erosion of biodiversity is that there is an underlying disparity between the private and social costs and benefits of biodiversity use and conservation (Dixon and Sherman, 1990; Perrings and Pearce, 1992). Private costs and benefits refer to those losses and gains as perceived by the immediate user of the environment: the farmer, the industrialist, the consumer. Social costs and benefits refer to the losses and gains that accrue to society as a whole. Social and private interests often do not coincide: what is good for me as an individual may impose costs on the rest of society — so-called `externalities'. Sometimes, as we will show, what is good for the world as a whole is also good for me as an individual, but no institutions exist for me to capture this `global value'. So, from the perspective of the individual (the farmer, the industrialist, the fisherman etc) it pays to destroy biodiversity. But from the point of view of society as a whole, it often pays to seek ways of sustainably utilizing that biodiversity and, on many occasions, it pays to protect it in some outright fashion. `Society' in this respect can be the local society, the nation, or the world as a whole.

Why do private and social interests diverge? Chapter 4 looks at this issue in detail, but we summarize the main factors here:

- Freely functioning markets are based on narrow self-interest. The upstream polluter has no incentive to account for the costs he imposes on a downstream user of the river. This is *market failure* and what is happening is that the downstream *externality*— the third party cost — is being ignored by the upstream polluter. Note that the failure arises from the free functioning of the market place. The economic approach should not be confused with any advocacy of free markets. However, as we shall see, functioning markets that are controlled in a particular way are powerful weapons for biodiversity conservation.
- Governments have a habit of intervening in markets. They may do this with the best of intentions. Indeed, they often intervene to remove the main elements of the externality caused by market behaviour. This is exactly what environmental regulation does. But, unfortunately, a great many other interventions are contrary to the interests of the environment, even where those interventions appear to serve some social purpose. Notable examples include activities to give financial incentives for deforestation, the underpricing of water resources, agricultural protection, and so on. There is therefore *intervention failure.* Note also that this failure can `co-exist' with market failure: they are not exclusive.
- Many conservation activities yield *global benefits.* If biodiversity is conserved in a tropical forest, for example, it yields a benefit to people in other countries, either because they simply want it to be there, or because it helps sustain basic biogeochemical cycles on which human survival depends. But if the country in question receives no financial or other resources to pay for these *global external benefits,* it will have no incentive to look after the biological resources. There is another form of market failure which we will call *global appropriation failure.* Note that this failure arises not from the functioning of markets, but from the fact that the markets are not there at all. They are *missing markets.* Moreover, as we show in Chapter 4, global missing markets can be present with local market failure and with intervention failure. When they are combined in this way they do much to explain why biodiversity disappears.

But how are we to demonstrate that these `failures' are important? They might, after all, be trivial when compared to the urgent need to change land uses for the benefit of economic development. The issue of *biodiversity resource valuation* is therefore a key issue in investment decisions about land use. Although the most sophisticated approaches are still largely restricted to developed country studies, those conducted elsewhere are increasingly demonstrating the magnitude and potential returns possible from sustainable use of biodiversity (SUB). The benefits of SUB should not be

exaggerated; some wild claims have been made for the economic value of plant-based pharmaceuticals, eco-tourism and so on. Nonetheless, it is important to tease out what these values might be, and to explain why they are not being more widely realized.

THE PROXIMATE CAUSES OF BIODIVERSITY LOSS

We distinguish between the proximate causes and the fundamental causes of biodiversity loss. Proximate causes show up as the more popular explanations of biodiversity loss: eg logging, agricultural clearance of forested land, pollution. Fundamental causes lie behind these proximate causes and are rooted in economic, institutional and social factors. The main proximate cause of loss is *land conversion,* ie the conversion from one land use to another, where land use includes sustainable management systems or even doing nothing with the land at all (wilderness). Table 2.1 shows some rates of conversion of natural habitats to agriculture.

Table 2.1 *Conversion of natural habitat to agriculture*

	1900 million hectares pa cropland	1980 million hectares pa cropland	% change
Developing			
Sub-S Africa	73	222	+204
Latin America	33	142	+330
South Asia	89	210	+136
China	89	134	+51
South-east Asia	15	55	+267
Developed			
North America	133	203	+53
Europe	145	137	-5
(ex) USSR	147	233	+58

Source: International Institute for Environment and Development and World Resources Institute (1987)

Table 2.2 *Recent rates of conversion to specialized agriculture*

	Conversion to cropland: Increase in area under crops 1977/9–1987/9 (%)			Conversion to pastureland: Increase in area under permanent pasture 1977/9–1987/9 (%)	
1	Suriname	53	1	Korea Rep.	116
2	Paraguay	47	2	Ecuador	44
3	Burkina Faso	28	3	Paraguay	33
4	Thailand	22	4	Canada	27
5	C d'Ivoire	21	5	Thailand	27
6	Uganda	20	6	Costa Rica	24
7	Mongolia	19	7	Philippines	23
8	Guinea Bissau	18	8	Vietnam	21
9	Brazil	17			
10	Rwanda	17			

Source: World Resources Institute and International Institute for Environment and Development (1992)

High rates of conversion have clear implications for biodiversity loss. Of special concern are rates of conversion in the so-called megadiverse states: areas identified as being of high species endemism. Table 2.2 shows historical conversion rates to specialized agriculture.

THE ECONOMIC VALUATION OF ENVIRONMENTAL GOODS

A common taxonomy for environmental asset valuation is presented in Table 2.3. Conceptually, total economic value (TEV) of an environmental resource consists of its use value (UV) and nonuse value (NUV). A use value is much as it sounds — a value arising from an actual use made of a given resource. This might be the use of a forest for timber, or of a wetland for recreation or fishing, and so on. Use values are further divided into direct use values (DUV), which refer to actual uses such as fishing, timber extraction etc; indirect use values (IUV), which refer to the benefits deriving from ecosystem functions such as a forest's function in protecting the watershed; and option values (OV), which is a value approximating an individual's willingness to pay to safeguard an asset for the option of using it at a future date. This is like an insurance value.

Table 2.3 *Categories of economic values attributed to environmental assets*

	Use values	Total economic value	Non-use values	
Direct use	*Indirect use*	*Option values*	*Bequest values*	*Existence values*
Outputs directly consumable	Functional benefits	Future direct and indirect values	Use and Non-use value of environmental legacy	Value from knowledge of continued existence
Food, Biomass, Recreation, Health	Flood control, Storm protection, Nutrient cycles	Biodiversity, Conserved habitats	Habitats, prevention of irreversible change	Habitats, Species, Genetic, Ecosystem

Non-use values (NUV) are slightly more problematic in definition and estimation, but are usually divided between a bequest value (BV) and an existence or `passive' use value (XV) (see Arrow et al, 1993). The former measures the benefit accruing to any individual from the knowledge that others might benefit from a resource in future. The latter are unrelated to current use or option values, deriving simply from the existence of any particular asset. An individual's concern to protect, say, the blue whale although he or she has never seen one and is never likely to, could be an example of existence value (see for example Randall and Stoll, 1983). Thus in total we have:

$$TEV = UV + NUV = (DUV + IUV + OV) + (XV + BV)$$

A sizeable literature has built up around the application of valuation techniques to a range of biological resources (see Chapter 6). As an example, Table 2.4 shows a set of identified functions of forests. Although not indicating the value category of each function, it is fairly clear where most belong, and also the extent of overlap in valuation. Thus direct use of timber and an indirect watershed protection function are often mutually exclusive, and double counting should therefore be avoided.

How useful the TEV classification is in practice is debatable. Existing valuation techniques can distinguish use values from `non-use' values, but attempts to isolate option, bequest and existence value are more problematic. Following recent legislative and legal controversies in the US and Australia, the concept of existence or `passive' use values has recently come under considerable scrutiny (Rosenthal and Nelson, 1992; NOAA 1992; Desvousges et al, 1993).

Table 2.4 *Environmental functions of forests*

Sources of materials and services	Sink for wastes	General and life support
Timber	Absorption of waste	Genetic pool
Fuelwood	Recycling nutrients	Climate regulation
Other business products	Watershed protection	Carbon fixing
Non-wood products	Protecting soil quality and	Habitat for people, flora and
Agricultural production	erosion resistance	fauna
Recreation and tourism		Aesthetic, cultural and spiritual source
		Scientific data

The claim that existence value in some sense represent `counter-preferential' values, motivated by moral concern, obligation or altruism, clearly poses problems for the conventional definition of an economic value. However, by taking the purpose of benefit measurement to be one of demonstrating economic value, however it is motivated, many of these problems disappear. Nonetheless, it is as well to be aware that the underlying principles and procedures for economic valuation are still debated.

IS TOTAL ECONOMIC VALUE REALLY TOTAL?

It is tempting to think that economists have captured all there is to know about economic value in the concept of TEV. But this is obviously not correct. First, recall that they are not claiming to have captured *all* values, merely economic values. Second, many ecologists say that total economic value is still not the whole economic story. There are some underlying functions of ecological systems which are prior to the ecological functions that we have been discussing (watershed protection and so on). Turner (1992) calls them `primary values'. They are essentially the system characteristics upon which all ecological functions are contingent. There cannot be a watershed protection function but for the underlying value of the system as a whole. There is, in some sense, a `glue' that holds everything together, and that glue has economic value. If this is true, then there is a total value to an ecosystem or ecological process which exceeds the sum of the values of the individual functions.

The discussion suggests three reasons why biological diversity is important.

1 The first reason is based on the concept of economic value. If biodiversity is economically important we would expect this to show up in *expressed willingness to pay for its conservation*. Shortly, we will show that this is indeed the case.

2 Economic value measurement will understate `true' economic value because of the probable failure to measure *primary life support functions*. This kind of economic value is difficult to observe because it is unlikely to be recognized until some disastrous event has happened: landslides consequent upon deforestation, loss of fishing grounds due to pollution, and so on.

3 Economic value does not capture — nor is it designed to capture — *intrinsic value*.

3 CONSERVATION VERSUS DEVELOPMENT

THE COSTS AND BENEFITS OF LAND USE CONVERSION

The individual's view

Land use conversion is the primary factor explaining biodiversity loss. It is essential, then, to understand why land use conversion takes place. We begin with a stylized situation: a rational economic agent deciding whether to conserve or develop the land he owns or rents. To `fix' the context we might assume that the decision in question is whether to conserve an area of tropical forest or to develop it, say by clearing it for agriculture. Economic rationality suggests that this

decision will be determined by the relative profitability, or *rate of return*, of the two options. Within the conservation option we include *sustainable use* of the forest for, say, agro-forestry or for non-timber products such as medicinal plants, eco-tourism etc. Note that the relevant rates of return are those that accrue to the land owner or tenant. At this point no account is taken of any returns to society or to the world as a whole.

In this simplified situation, then, the decision to conserve or use sustainably will be the right one if:

Rate of return from SUB > Rate of return from `Development'

where SUB is the `sustainable use of biological diversity', ie the conservation option. This can be written:

$$B(SUB) — C(SUB) > B(DEV) — C(DEV) \quad (1)$$

or

$$B(SUB) — C(SUB) — [B(DEV) — C(DEV)] > 0 \quad (2)$$

where

B(SUB) = the benefits of sustainable use of the forest;

B(DEV) = the benefits of traditional development of the land for, say, agriculture or forestry or industry;

C(SUB) = the costs of the sustainable use option;

C(DEV) = the costs of the development option.

Rule 1 or 2 simply says that the net benefits from sustainable use of biodiversity should exceed the net benefits from development if conservation is to be preferred to development. Recall that the benefits and costs here are all defined in terms of the returns to the individual. They are *private costs and benefits*.

One complication that needs to be addressed immediately is the relevance of time. Typically, the land owner will prefer benefits now rather than later, and costs later rather than now. To allow for this we have to introduce *iscounting*. Discounting permits us to compare gains and losses that occur over different time periods. The simplest way to approach it is to imagine the choice of $1 now or $1 next year. From the individual's standpoint there are several reasons why the $1 is preferred now. First, the $1 now could be invested (in a bank or in land or other assets) to become $1 plus accrued interest next year. If the interest rate is r, then the $1 now is worth $(1+r) next year. So the comparison of $1 now and $1 next year becomes a comparison of $1 next year and $(1+r) next year. Clearly, the $(1+r) is preferred, which is the same as saying that $1 now is preferred. Another way of putting this is that $1 next year is worth $1/(1+r) now. This sum is the *present value* of the future flow of income, the factor 1/(1+r) is the *discount factor* and r is the *discount rate*. Note that on this analysis the discount rate can be found by seeing what rate of return can be earned by investing money. This is known as the *opportunity cost of capital* approach to discounting. But there are other reasons for discounting. First, the individual may simply be impatient, even if he or she knows that they will be alive next year and even if they know the $1 will certainly be there next year. This impatience motive is called *pure time preference* discounting. As it happens, this time preference may be reinforced by the fact that the individual is not certain of being alive in the following year. There will be some discounting to reflect mortality risk. This is usually regarded as being part of pure time preference. A second factor is that the individual may expect to be richer next year anyway, so that $1 of additional income next year does not appear to be quite so important as $1 now. This is *discounting due to diminishing marginal utility of income*, where `utility' here is simply the economist's term for individual welfare or well-being, and `marginal' means `extra'. So this form of discounting simply reflects the expectation that an extra (marginal) $1 next year is worth less in utility terms than an extra $1 now. The resulting discount rate that emerges from these considerations, s, is known as the *social time preference rate*. The rates s and r tend to differ in the real world and there is a debate as to which one should be used. For our purposes we need to note only that, from the individual's standpoint, both s and r are positive, ie s > 0 and r > 0.

Allowing for time, rule (2) needs to be restated in terms of *present values* to be:

$$PV[B(SUB) — C(SUB)] — PV[B(DEV) — C(DEV)] > 0 \quad (3)$$

where PV(B) = $\sum B_t/(1+r)^t$, or $\sum B_t/(1+s)^t$, and similarly for costs.

Equation (3) already gives us an insight to what is happening with land use conversion. If the value of SUB is low and that for DEV is high, then, other things being equal, land conversion will take place. Very simply, from the standpoint of the individual, it is more profitable to `develop' than to conserve. Now, if the benefits of SUB accrue in unmarketed form — ie there is no obvious market for them — then the individual landowner has no incentive to take account of them. They may, for example, accrue to other people, even other people in some other country (like existence value). The benefits from development are `real' and tangible. The benefits from SUB are often intangible. Put another way, the rate of return from conservation will appear to be very low, even zero, to the landowner or tenant.

The effect of discounting is a little more complex but worth exploring. A great many land conversions, especially those that relate to tropical forests, result in unsustainable land uses. Thus, a forest area may be cleared through burning. The burning converts the nutrient matter in the forest biomass into ash which then acts as a fertilizer for the nutrient-weak forest soil. The fire also acts as a pesticide. Crops are then grown and these use up much of the stock of nutrients in the soil. So, what was a renewable resource system becomes a non-renewable resource system: the living forest is replaced by a `mine' of nutrients. As the nutrient value of the soil declines, so the land user reaps lower and lower returns from staying on the land. He eventually moves on and clears the next area of land. Left alone, new forest might be generated. But other users may come into the area, say a rancher with cattle. These cattle then use up the remaining store of nutrients in the soil and then the land may become truly degraded, ie unsuited to any agricultural use and with low prospect of regenerating forest.

The alternative to forest clearance is sustainable use of the land. If it is truly sustainable (some such allegedly sustainable uses themselves damage the forest), then that use can last for a very long time. This suggests that it should be preferable to the returns to `nutrient mining' discussed above, since that ends when the soil is truly depleted. *But the problem is that discounting can make the non-sustainable use preferable to the sustainable use.* Figure 3.1 shows why this might be so. Because the future benefits from sustainable use are discounted, they may appear to matter little to the land owner, whereas the immediate benefits of conversion are higher, even though they result in zero long-term benefits. Of course, if the land-owner knows there is some further forest area he can colonize once the existing one is depleted, this adds to the incentive to deforest.

Figure 3.1 (a) shows the hypothetical `cash flows' for non-sustainable nutrient mining and for sustainable land use. Nutrient mining has the higher short-term profits but zero long-term profits. Sustainable land use has long-term profits but lower immediate returns. Figure 3.1 (b) shows the effect of discounting. Both lines `bend downwards'. The present values are the areas under these curves. If the heavy shaded area is greater than the hatched area, then the non-sustainable use is preferred.

Thus, two factors already provide some insight into why deforestation — and hence biodiversity loss — appears to be `economically rational' from the individual's standpoint. First, the returns from clearance may simply be higher than the returns from conservation because the latter may consist of non-market benefits or benefits that accrue to people other than the landowner. Second, the effect of discounting is to discriminate against sustainable uses of the land if those uses have lower *initial* returns, even though the returns last much longer.

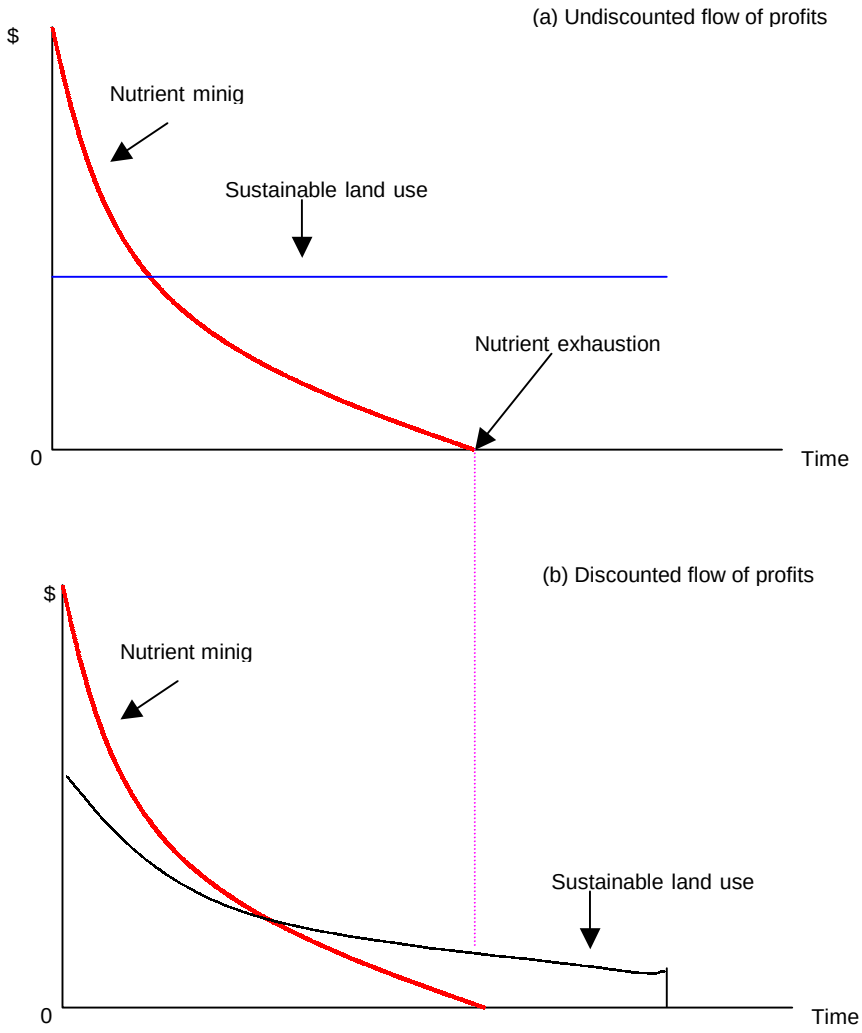
Society's view and the world view

The analysis so far has looked at land conversion from the individual's standpoint. What of society's view? From the social standpoint we need to redefine the benefits and costs in equation (3). We also need to ask if society's discount rate is the same as that of the individual.

As Chapter 2 showed, the benefits of conserving biodiversity are divided typically into use values and non-use values. Together, use and non-use values make up the total economic value of biodiversity conservation.

The second adjustment arises from the fact that, as noted above, both use and non-use values can reside in the host nation or globally (where globally means all nations other than the host nation). Using `n' to denote national and `g' to denote global, and using the TEV notation from Chapter 1, we can now write:

Figure 3.1 Discounting and sustainable land use



$$TEV(SUB) = UV + NUV = DUV + IUV + OV + BV + XV \qquad (4)$$

and

$$TEV(SUB) = UV_n + UV_g + OV_n + OV_g + BV_n + BV_g + XV_n + XV_g \qquad (5)$$

The expression for the cost-benefit rule, then, is that sustainable use will be preferred if:

$$PV\,[TEV(SUB) — C(SUB)] — PV[B(DEV) — C(DEV)] > O \qquad (6)$$

Equation (6) sets the requirements for the comparison of sustainable land use and its opportunity cost, namely the forgone development values. It indicates what would be needed for sustainable use to be preferred over traditional development land use if a national host country standpoint is taken, and if that country seeks to secure the biggest gains in national efficiency. It tells us that conservation is preferred if the *national* gains are greater than the costs, and that those national gains will be larger still if the country can `capture' some of the global use and non-use values. Once again, if the individual land user does not get part of the national gains from conservation, or part of the global gains, then he has no incentive to act in accordance with equation (6). He will simply operate according to his own private gains and losses.

This divergence between social, global and private returns does much to explain why biodiversity is being reduced. Chapter 4 takes the analysis further.

MEASURING BENEFITS AND COSTS

Equations (3) and (6) are the fundamental equations needed to establish *why* biodiversity loss occurs. If social, global and private costs and benefits diverge then there will be a strong incentive to convert land. This is market failure in both the local and global sense of Chapter 2. As we shall see in Chapter 4, government intervention actually makes the situation worse because it often exaggerates the private returns from destroying biodiversity.

The rates of return are measured in terms of economic benefits and economic costs. These costs and benefits have specific definitions in economics. Benefits and costs reflect either *willingness to pay* (WTP) to secure a gain (or benefit) or to avoid damage (a cost); or *willingness to accept* (WTA) compensation to forgo a gain or tolerate a cost. These WTP and WTA measures in turn reflect individuals' preferences which are the `raw material' of economic valuation. If, for some reason, it is thought right to reject individual preferences as the basis for resource allocation, then the economic efficiency criterion will not be appropriate. Table 3.1 summarizes the relevant linkages.

Whereas it was traditionally thought that WTP and WTA would not diverge very much, recent evidence suggests that they can, and do. Typically, WTA may be a magnitude several times that of WTP, a difference unaccounted for by the limits set by income on WTP and which limit is not present for WTA. Differences are explained mainly in terms of (a) loss aversion, ie valuations of damages (a loss) are higher than the comparable gain relative to some initial endowment of assets, including environmental assets; and (b) limited substitutes for environmental goods (Kahneman and Tversky, 1979; Knetsch and Sinden, 1984; Knetsch, 1989; Hanemann, 1991). The relevance of the WTP/WTA distinction is that it will matter which measure is used when comparing the rates of return to alternative land uses. This issue is considered further when appraising the evidence of different rates of return. In the meantime, Table 3.2 illustrates the degree of divergence found between WTA and WTP in valuation studies.

Table 3.1 *The sources of economic value*

Gains	Losses
Preference for securing a benefit = = >	Preferences against losses = = >
Willingness to pay for a benefit or	Willingness to pay to avoid a loss or
Willingness to accept compensation to forgo the benefit	Willingness to accept compensation to tolerate a loss

ECONOMIC VALUES AND MORAL ISSUES

Many people feel it is quite wrong to allow economic values to play any role in determining what should and should not be `saved' by way of conservation. For the implication of equation (6) is that if the total economic value of conservation using a WTP metric does *not* prove to be greater than the value of development, then development `should' take place. They would argue that conservation is a moral issue, to be determined by some discussion of the `rights' of other species, the rights of indigenous peoples and other minorities, and by our moral obligations to future generations. Status of biodiversity is an end in itself rather than an instrumental means to an end. If the economic approach puts conditions on absolute or permanent protection then such rights are contravened and the paradigm is unacceptable.

The idea that the `moral' view is opposed to the `economic' view rests on may confusions. First, the economic view is itself a moral view — it takes what is effectively a *utilitarian* approach to conservation. What the critics are complaining of is not so much the economics as the underlying philosophy of normative economics, *utilitarianism*. Of course, it is quite proper for such a philosophical debate to take place. The problem is that, in the absence of `metaethical' principles, principles that enable us to choose between apparently competing philosophies, the debate risks being rather sterile *from the standpoint of getting things done*. Put another way, the moral debate has gone on for a very long time and is as relevant to, say, crime and punishment as it is to biodiversity conservation. The fact that such debates have not been resolved is not surprising, but, of course, that in turn cannot be a reason for not continuing to try and resolve it. The problem is that much conservation policy to date has been based on non-utilitarian approaches. Yet by many accounts the current situation is one of crisis. It would seem fair, then, to choose between the competing philosophies according to their

potential for saving biodiversity in real world contexts. We argue that this favours the economic-utilitarian approach.

Table 3.2 *The divergence between willingness to pay and willingness to accept (US$)*

Study and entitlement	Means			Medians		
	WTP	WTA	Ratio	WTP	WTA	Ratio
Hypothetical surveys:						
Hammack and Brown (1974): marshes	$245	$1.0	4.2			
Sinclair (1978): fishing	35	100	2.9			
Banford et al (1979)	43	120	2.8	47	129	2.7
Fishing Pier						
Postal Service	22	93	4.2	22	106	4.8
Bishop and Heberlein (1970): goose hunting permits	21	101	4.8			
Rowan et al (1980): visibility	1.33	3.49	2.6			
Brookshire et al (1980): elk hunting*	54	143	2.6			
Heberlein and Bishop (1985): deer hunting	31	516	16.5			
Real exchange experiments:						
Knetsch and Sinden (1984): lottery tickets	1.28	5.18	4.0			
Heberlein and Bishop (1985): deer hunting	25	172	6.9	1.33	3.40	2.6
Coursey et al (1987): taste of sucrose octa-acetate		3.45	4.17	6.9		
Brookshire and Coursey (1987): park trees	10.12	417	1.4	1.35	12.96	2.1

Source: Kahneman, Knetsch and Thaler (1990)

Alternative moral standpoints would also be more tenable if they confronted the real world context of making choices. If all biological resources have `rights' to existence then presumably it is not possible to choose between the extinction of one set of them rather than another. All losses become morally wrong. But biodiversity loss proceeds apace for the reasons we have cited and for one other we have not so far mentioned: the competition between mankind and other species for the available space. The reality is that little can be done to prevent huge increases in the world's population — it is in that respect `too late' for a good deal of the world's biological diversity. If so, it is essential to choose between different areas of policy intervention — not everything can be saved. This view is reinforced by the fact that the world is extremely unlikely to devote major resources to biodiversity conservation. We can argue that it should, but we know it will not. The issue then becomes one of using the existing budgets as wisely as possible. If not everything can be saved then a *ranking* procedure is required. And such a ranking is not consistent with arguing that everything has a right to exist.

Moreover, if we are right, and economic `causes' are very important, then, presumably, the moral view would sanction the correction of the economic factors giving rise to excess biodiversity loss. That would be a start at least. In other words, whatever moral standpoint is taken it does not affect the design of a practical agenda for conservation, and that agenda should begin with the economic factors.

Finally, even if some do not like the economic-utilitarian approach, it has a major function which is not served by any other approach to conservation. It explains why biodiversity is being lost. It tells us that, since people very often *are* utilitarian in their decisions about land use and conservation, a utilitarian approach is needed in order to understand the process of loss, and hence the process of policy correction.

* Middle-level several used in study

4 THE CAUSES OF BIODIVERSITY LOSS

ECONOMIC FAILURE

Chapters 2 and 3 introduced the idea of `economic failure': the inability of existing markets to capture the `true' value of natural resources. Two sources of such failure were identified:

1 *Market failure* — distortions due to the `missing markets' in the external benefits generated by biodiversity conservation; and

2 *Intervention or government failure* — distortions due to government actions in intervening in the workings of the market place.

Within market failure we distinguished *local market failure* and *global market failure.* The former relates to the inability of markets to capture some of the local, national benefits of biodiversity conservation. Or, looked at from the standpoint of land conversion, local market failure refers to the failure of markets to account for the external costs of biodiversity loss because of land conversion. The latter concept — global market failure — relates to the fact that biodiversity conservation yields external benefits to people outside the boundaries of the nation faced with the development/conservation choice.

All these forms of failure can co-exist. Moreover, they exist very often in a context of rapidly changing population as far as developing countries are concerned. But these forms of failure are not peculiar to developing countries. Rich countries also have local market failure and government failure as well. They may have less global failure because the biodiversity that has global value is often, but not always, in developing or transition economies. Figure 4.1 summarizes the situation.

Figure 4.1. *Categories of economic failure*

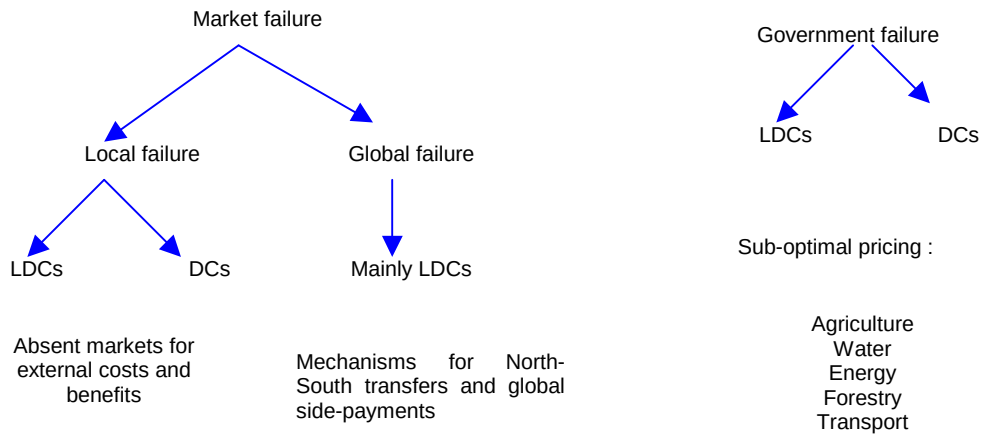


Figure 4.2 introduces a diagrammatic exposition of the types of economic failure. The horizontal axis shows the amount of land converted to, say, agriculture. The vertical axis shows money. The downward sloping line MPB_i is the `marginal private benefits' of land conversion, ie the extra revenue obtained by the farmer by converting the land from forest to agriculture. The line $MC_i$ is the marginal cost to the farmer of making the conversion. The `rational farmer' will equate MC_i and MPB_i in order to maximise profits.[1] Hence the amount of land conversion that actually takes place is L_P.

[1] To see this, profits, π, equal PB(L) — C(L), ie the private revenues from conversion less the costs of conversion. Maximizing profits and differentiating gives

$$d\pi/dL = dPB/dL — dC/dL = 0$$

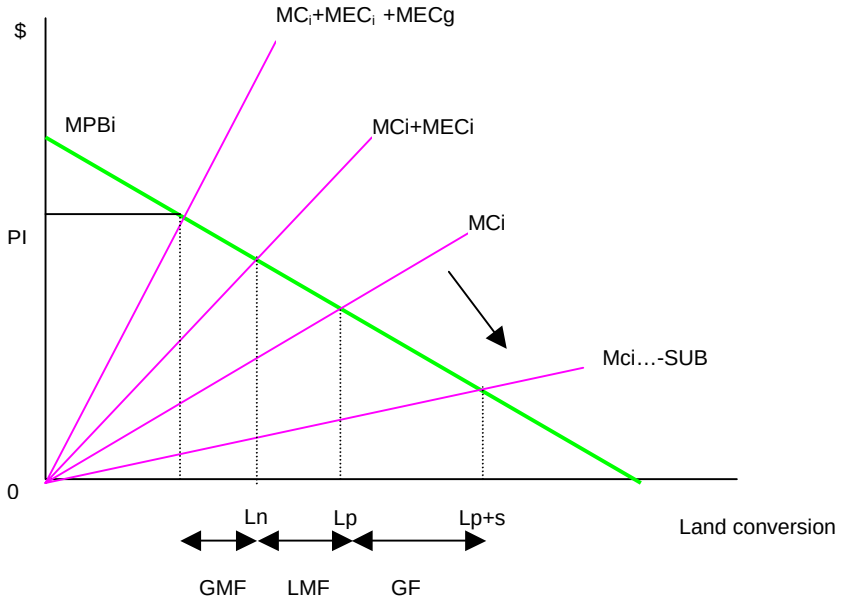
or

$$dPB/dL = dC/dL$$

Now suppose the farmer is subsidized to convert the land. The effect can be shown as a lowering of MC_i to MC_i — SUB, where SUB refers to the subsidy. That is, private costs are lowered. This induces the landowner to expand the level of land conversion to L_{P+S}. The distance L_P — L_{P+S} is a measure of *government failure* (GF).

What should the level of land conversion be? To find this we need to estimate the value of the two externality components: the local and global externality. This involves *valuation.* If we know the value of the damage done to the nation from such land conversion — eg lost indirect and direct use values — then some estimate of the local external cost can be made. The diagram shows this as MEC_i, ie the marginal external cost imposed on the nation. If this externality is `internalized', ie if the farmer is made to account for it in some way (eg by taxation or by bearing higher costs because the land is zoned for conservation) then the relevant `optimum' moves to L_n. Note that L_n is less than L_P, so that internalizing the externality involves less land conversion and hence more biodiversity conservation. The distance L_n — L_P is a measure of the *local market failure* (LMF).

Figure 4.2 *Measuring economic failure*



The same procedure can be used to account for the global externality, the value of the losses borne by people outside the nation that owns the forest. The distance L_G — L_N is a measure of *global market failure* or *global appropriation failure* (GMF).

The analysis also provides us with a rule for the `proper pricing' of land. It is given by:

$$P_L = MC_i + MEC_i + MEC_g$$

and this is shown on the diagram. In other words, if land could be priced at the cost of its conversion *plus* the external costs of conversion, then the amount of conversion would be economically `optimal'. Note that this optimal amount of land conversion is not zero — some deforestation still takes place. This result of the economic analysis is often regarded by environmentalists as unsatisfactory. Indeed, if one adopts the `moral' standpoint discussed at the end of Chapter 3 it will appear to be `immoral' to allow *any* deforestation to take place. This illustrates a further contrast between the economic and the moral standpoint: the latter tends to focus on the costs of deforestation *only.* It ignores the benefits of deforestation, ie the gains to be obtained by the farmer in question. The economic approach quite explicitly compares these benefits with the costs.

How does population growth and economic growth fit into the picture in Figure 4.2? It can be illustrated by shifting the MPB curve to the right over time. If the costs of further land conversion do not change (which they might as the `frontier' gets further and further from established urban centres),

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But the left hand expression is marginal private benefits (MPB_i) and the right hand expression is marginal cost (MC_i).

then such shifts will make it more worthwhile to convert land. The reader can experiment with the diagram to see that all the `optima' move to the right if the MPB curve shifts outwards.

Clearly, then, if there is to be a policy on biodiversity conservation it has to focus on the main fundamental causes of loss identified in this section:

- local market failure which will need to be addressed by local measures such as the zoning of land to forbid, say, forest burning;
- global market failure which will need to be addressed by `creating' markets in global value and ensuring that compensation for forgoing the development option is paid to the landowner;
- intervention failure which will need to be addressed by showing the government that there are gains to be made by avoiding expensive policies of subsidization of forest clearance;
- limitation of population growth.

ILLUSTRATING ECONOMIC FAILURE: INTERVENTION FAILURE

Examples of intervention failure are, by now, well known (Pearce and Warford, 1993) and include the subsidies to forest conversion for livestock in Brazil up to the end of the 1980s; the subsidies to beef in Botswana, inflated by preferential tariffs in the European Community; hedgerow removal and over-intensive farming arising from above-equilibrium guaranteed prices under the European Common Agricultural Policy; the under-pricing of irrigation water whether in California or Pakistan, and so on. What government intervention does is to distort the competitive playing field. We are used to hearing businesspeople adopt this language, but, while they often see environmental regulations as the means of hampering their competitive efficiency, the truth is that the same argument shows powerfully why the conservation of biological diversity is an uphill struggle. Governments effectively subsidise the rate of return to land conversion, tilting the economic balance against conservation.

Table 4.1 assembles some information on the scale of the distortions that governments introduce. Such distortions are widespread. While some OECD countries tax their agricultural sectors, most subsidize agriculture. The extent of these subsidies is measured by the Producer Subsidy Equivalent (PSE)[2] which indicates the proportion of revenues farmers derive from various price support mechanisms. Table 4.1a indicates how for much of the '80s overall support has *increased* considerably. The aggregate figures disguise marked differences between farm products, with some arable crops benefiting from support in excess of 90 per cent. In the developing world, agricultural prices tend to be kept *below* their comparable border price for distributional reasons. On the other hand *input prices* for products such as pesticides and fertilizers are frequently subsidized, to the detriment of traditional integrated pest management systems. The effects of excessive use on mortality and related ecological problems have been well documented (Repetto, 1985).

Table 4.1a *OECD agricultural subsidies*

	Producer subsidy equivalents		
	1981–84	*1985–88*	*1989–92*
Australia	11	12	12
Canada	30	47	45
EC	32	47	46

[2] 2 A PSE equals:

$$Q.(P-PWnc) + DP - LV + OS$$

and a percentage PSE equals:

$$PSE/(Q.P + DP - LV)$$

where Q = volume of production; P = domestic producer price; PWnc = border price in national currency; DP = direct payments; LV = levies on production; OS = other support.

	Producer subsidy equivalents		
	1981–84	*1985–88*	*1989–92*
Japan	63	74	68
Sweden	38	55	57
United States	27	35	27
OECD	33	46	43

Source: OECD (1993)

Table 4.1b *Agriculturarl input subsidies in developing countries*

	Irrigation: *Ratio of prices to capital + operating costs*	Pesticides: *Subsidy as a % of retail cost*
Bangladesh	18*	
Indonesia	14	82
Rep of Korea	18	
Nepal	7	
Philippines	22	
Thailand	5	
China		19
Colombia		44
Ecuador		41
Egypt		83
Ghana		67
Senegal		89

Source: Pearce and Warford (1993)

Intervention failure is not confined to agriculture. Other notable sectors include commercial energy (Kosmo, 1989), where support may be costly both environmentally and in terms of diverted government revenues. The effects of regional incentives are sometimes less apparent but no less damaging. Historic use of investment, tax and credit incentives, as well as subsidized infrastructure, in the Brazilian Amazon have all raised the return to agriculture relative to forest conservation. Elsewhere, interventions which artificially inflate financial returns to timber production have the same effect. Most notable are the use of foreign investment tax breaks and low royalties or concession fees.

ILLUSTRATING ECONOMIC FAILURE: GLOBAL APPROPRIATION FAILURE

We can illustrate the phenomenon of `missing markets' with two examples which are highly relevant to biodiversity: the `non-use' or `existence' value possessed by individuals in one country for wildlife and habitat in other countries, and the `indirect use' carbon storage values of tropical forests. Global appropriation failure arises because these values are not easily captured or appropriated by the countries in possession of biological resources.

Non-use values

Economists use methods of measuring individual preferences, as revealed through individuals' `willingness to pay' to conserve biodiversity. The methodologies include *contingent valuation* (CVM),

* Operating costs only

which functions through sophisticated questionnaires which ask people their willingness to pay, and other techniques such as the *travel cost method,* the *hedonic property price* approach and the *production function* approach — see Chapter 5. The economic values that are captured in this way are likely to be a mix of potential use and non-use values. Use values relate to the valuation placed on the resource because the respondent makes use of it or might wish to make use of it in the future. Non-use values, or `passive use values' as they are also called, relate to positive willingness to pay even if the respondent makes no use of the resource and has no intention of making use of it.

`Global valuations' of this kind are still few and far between. Table 4.2 assembles the results of CVMs in several countries. These report willingness to pay for species and habitat conservation in the respondents' own countries. These studies remain controversial, especially in light of the findings of the recent `blue ribbon' panel on contingent valuation in the US (Arrow et al, 1993), although that same panel basically gave CVM a good bill of health provided rigorous rules of investigation are pursued. While we cannot say that similar kinds of expressed values will arise for protection of biodiversity in other countries, even a benchmark figure of, say, $10 pa per person for the rich countries of Europe and North America would produce a fund of $4 billion pa. This is around four times the mooted size of the fund that will be available to the Global Environment facility in its operational phase as the financial mechanism under the two Rio Conventions and in the context of its continuing role in capturing global values from the international waters (see Chapter 8), and perhaps 10 times what the Fund will have available for helping with biodiversity conservation under the Rio Convention.

Table 4.2 *Preference valuations for endangered species and prized habitats*

Species		Preference valuations (US 1990 $ pa per person)
Norway:	brown bear, wolf and wolverine	15.0
USA:	bald eagle	12.4
	emerald shiner	4.5
	grizzly bear	18.5
	bighom sheep	8.6
	whooping crane	1.2
	blue whale	9.3
	bottlenose dolphin	7.0
	California sea otter	8.1
	Northern elephant seal	8.1
	humpback whales[1]	40–48 (without information)
		49–64 (with information)
Habitat		
USA:	Grand Canyon (visibility)	27.0
	Colorado wilderness	9.3–21.2
Australia:	Nadgee Nature Reserve NSW	28.1
	Kakadu Conservation	40.0 (minor damage)
	Zone, NT[2]	93.0 (major damage)
UK:	nature reserves[3]	40.0 (`experts' only)
Norway:	conservation of rivers	59.0–107.0

Notes: (1) respondents divided into two groups, one of which was given video information; (2) two scenarios of mining development damage were given to respondents; (3) survey of informed `expert' individuals only.

Source: Pearce (1993)

Carbon storage

All forests store carbon so that, if cleared for agriculture, there will be a release of carbon dioxide which will contribute to accelerating the greenhouse effect and hence global warming. In order to derive a value for the `carbon credit' that should be ascribed to a tropical forest, we need to know (1)

the net carbon released when forests are converted to other uses, and (2) the economic value of one tonne of carbon released to the atmosphere.

Carbon will be released at different rates according to the method of clearance and subsequent land use. With burning there will be an immediate release of CO_2 into the atmosphere, and some of the remaining carbon will be locked in ash and charcoal which is resistant to decay. The slash not converted by fire into CO or charcoal and ash decays over time, releasing most of its carbon to the atmosphere within 10–20 years. Studies of tropical forests indicate that significant amounts of cleared vegetation become lumber, slash, charcoal and ash. The proportion differs for closed and open forests; the smaller stature and drier climate of open forests result in the combustion of higher proportion of the vegetation.

If tropical forested land is converted to pasture or permanent agriculture, then the amount of carbon stored in secondary vegetation is equivalent to the carbon content of the biomass of crops planted, or the grass grown on the pasture. If a secondary forest is allowed to grow, then carbon will accumulate, and maximum biomass density is attained after a relatively short time.

Table 4.3 illustrates the net carbon storage effects of land use conversion from tropical forests (closed primary, closed secondary, or open forests) to shifting cultivation, permanent agriculture, or pasture. The negative figures represent emissions of carbon; for example, conversion from closed primary forest to shifting agriculture results in a net loss of 194 tC/ha. The greatest loss of carbon involves change of land use from primary closed forest to permanent agriculture. These figures represent the once and for all change that will occur in carbon storage as a result of the various land use conversions.

Table 4.3 *Changes in carbon with land use conversion (tC/ha)*

	Original C	Shifting agriculture	Permanent agriculture	Pasture
Original C		79	63	63
Closed primary	283	-204	-220	-220
Closed secondary	194	-106	-152	-122
Open forest	115	-36	-52	-52

Shifting agriculture represents carbon in biomass and soils in second year of shifting cultivation cycle.

Source: Brown and Pearce (1994)

The data suggest that, allowing for the carbon fixed by subsequent land uses, carbon released from deforestation of secondary and primary tropical forest is of the order of 100–200 tonnes of carbon per hectare.[3]

The carbon released from burning tropical forests contributes to global warming, and we now have several estimates of the minimum economic damage done by global warming, leaving aside catastrophic events. Recent work by Fankhauser (1994) suggests a `central' value of $20 of damage for every tonne of carbon released. Applying this figure to the data in Table 4.3, we can conclude that converting an open forest to agriculture or pasture would result in global warming damage of, say, $600–1000 per hectare; conversion of closed secondary forest would cause damage of $2000–3000 per hectare; and conversion of primary forest to agriculture would give rise to damage of about $4000–4400 per hectare. Note that these estimates allow for carbon fixation in the subsequent land use.

How do these estimates relate to the development benefits of land use conversion? We can illustrate with respect to the Amazon region of Brazil. Schneider (1992) reports upper bound values of $300 per hectare for land in Rondonia. The figures suggest carbon credit values two to fifteen times the price of land in Rondonia. These `carbon credits' also compare favourably with the value of forest

[3] A refinement to these figures would estimate the present value of the carbon releases by discounting future carbon releases. That is, all the carbon is not released in the initial burning. There may be subsequent burnings, and there may be some slower release of carbon over time.

land for timber in, say, Indonesia, where estimates are of the order of $2000–2500 per hectare. All this suggests scope for a global bargain. The land is worth $300 per hectare to the forest colonist but several times this to the world at large. If the North can transfer a sum of money greater than $300 but less than the damage cost from global warming, there are mutual gains to be obtained.

Note that if the transfers did take place at, say, $500 per hectare, then the cost per tonne of carbon reduced is of the order of $5 tC ($500/100 tC/ha). These unit costs compare favourably with those achieved by carbon emission reduction policies through fossil fuel conversion. Avoiding deforestation becomes a legitimate and potentially important means of reducing global warming rates.

THE DISTRIBUTION OF CONSERVATION COSTS AND BENEFITS

Figure 4.2 demonstrated the economic rationality of land conversion when accounting for the global and national as well as private costs. Valuation of these externalities turns out to be a prerequisite for moving towards a `globally efficient' rate of conversion. In the case of forest conversion for example, values for carbon storage are more readily quantified than those of biodiversity. Uncertainty in benefit and cost accounting suggests that there cannot be a completely certain view of the optimal conversion level, nor what the optimal policy to reach it will look like. Economic valuation *can,* however, help move in the direction of an economically efficient or `first best' outcome. While this might make conservation policy seem like a hit and miss affair, greater precision will emerge with improved scientific understanding of ecological processes and a political appreciation of a latent global WTP for conservation.

Translating the economic rationality into effective national and local incentives is an equally challenging policy issue. As noted elsewhere, conservation in many countries is characterized by a spatial mismatch between costs and benefits (Wells, 1992a). In other words, economic benefits from conserved areas tend to be limited on a local scale, increase somewhat at a national level and, as is slowly becoming clear, can be substantial on a global scale. On the other hand, costs, in terms of forgone development benefits, tend to be locally significant and nationally and globally moderate. Appraising conservation policies in a cost-benefit framework, it is tempting to abstract from these distributional issues, providing an intervention increases aggregate welfare and gainers could *potentially* compensate the losers. The capacity for potential compensation is a common assumption in welfare economics, and economists term such welfare-generating changes `net welfare gains'. In the context of global conservation, the emphasis on potential rather than *actual* compensation is clearly misplaced. As incomes rise and pristine environments suffer greater degradation, global demand for conservation for ecotourism, recreation and other non-use benefits is likely to increase. Since many highly prized environments are in countries where conservation has high opportunity costs, meeting this global demand without compensation will accentuate the benefit mismatch. The problem is clear. A wider population of global beneficiaries from conservation habitually demonstrates a willingness to pay for conservation by visiting wildlife sites, subscribing to conservation groups or simply for the option of visiting or using indigenous resources. This demand restricts revenue opportunities for local populations who in effect subsidize the provision of a global public good.[4] The remedy would therefore seem straightforward. Either sufficient appropriable local benefits derive from, say, improved participation in conservation provision, or a system of international compensation is formulated. In an ideal world the latter would occur through a market for conservation goods. Such *global environmental markets* (GEMs), although historically `missing', are beginning to emerge albeit on a modest scale (Pearce, 1994 and Chapter 8). Public (official) ventures of this nature include disbursements under the Global Environment Facility, officially sanctioned debt-swaps and debt rescheduling. Private initiatives generally relate directly to existing legislation which price externalities and provide scope for mutually beneficial trades. Carbon off-set agreements are probably the clearest example (Faeth et al, 1994).

In the context of biodiversity conservation, most GEM transactions are at a national level, the exception being a number of private prospecting contracts for plant genetic materials (Gamez et al, 1993). What incentives these agreements provide at a household level is far from certain. Indeed, international transactions may be to limited effect if they cannot be applied and enforced where it matters most. Drawing on developed country experience, conservation easements, compensated set-

[4] In economics, the term *public good* refers to a good or service where consumption is `non-rival' and `non-exclusive'. This non-excludability characterizes benefits such as existence and option values.

aside and tradeable development rights have all been suggested as potential mechanisms for effecting workable agreements (Katzman and Cale, 1990; Cervigni, 1993a; Panayotou, 1994). The characteristics of these trades include appropriable mutual gains to participating landowners and sponsoring public or private organizations, and incentive compatibility. In other words, agreements must be structured so as to maximize the incentive to comply, while minimizing both the gains from cheating and policing costs at each level of administration and operation. While experience with such instruments is limited, not least by complex national and international legal implications, they are a promising attempt at bridging the gap (establishing channels) between benefit provision, global transfers and local appropriation.

5 METHODOLOGIES FOR ECONOMIC VALUATION

INTRODUCTION: VALUATION IN HIGH AND LOW INCOME CONTEXTS

Chapters 3 and 4 described the philosophy underlying the procedure for putting money values on preferences for environmental change. This chapter investigates the *methodologies* for eliciting economic values. A potentially important issue arises in that much of the world's threatened biological diversity is in the *developing world,* whereas the theory and practice of economic valuation has been developed and applied mainly in the *developed world.* Accordingly, it is important to assess whether rich country methodologies can be applied in poor country contexts. Prima facie, it could be argued that a number of the methodologies will not be applicable due to the absence of even moderately freely functioning markets for inputs (eg labour, capital, raw materials) and outputs (eg agricultural produce) in developing countries. In practice, there is a fairly extensive literature on the valuation of environmental change in developing countries and, by and large, the problems of application, while significant, are not insuperable.

It has been shown that procedures which require individuals to state their `willingness to pay' in hypothetical contexts for goods and services (contingent valuation), work well in most developing country contexts (Whittington et al, 1991; Boadu, 1992). As in any developed country, the respondent faces a budget constraint (income or wealth) and, provided he or she is familiar with the good being offered, responses in developing country contexts appear to be as reliable as in other contexts. Other techniques observe actual behaviour and infer valuations based on that behaviour. For example, if water users have three options facing them — water being brought to the door, water purchased from a local `kiosk', and getting water from a fairly distant well — it is possible to value the time spent collecting the well water by looking at the differences in prices in the three options (`discrete choice' approach). Additionally, approaches which link some change in an environmental variable to a change in a marketed output provide us with a potentially large set of estimates. This is, for example, how most studies on the economic costs of soil erosion have been carried out.

Extending these procedures to the valuation of biological diversity is complex. Indeed, the valuation of preferences for biodiversity is perhaps the most challenging issue in the context of economic valuation. But there are many use values, such as ecotourism, in which various valuation procedures (such as the travel cost method) might be used. Similarly, surveys measuring the forgone local use benefits from reserve designation or tourists' willingness to pay for park maintenance provide some guidance for conservation policy (Mercer et al, 1993; Moran, 1994). The focus of these studies is on the conservation of biological resources, but there may often be incidental diversity benefits if subjects are considered to be keystones or whole system pivots.

To date, procedures whereby individuals are asked their willingness to pay to conserve biodiversity *per se* have not been used in the developing country context. In developed countries, direct questioning on biodiversity preferences is largely limited to the preservation of well-known or `charismatic' species and ecosystems. Recent attempts to elicit preferences for less familiar biodiversity have encountered response difficulties when subject goods are difficult to explain or unknown to respondents, or where respondents lack experience of making similar transactions (Stevens et al, 1991; Hanley and Spash, 1993). The issue of whether value statements are the result of an information constraint or a more fundamental refusal to make implicit trade-offs, is yet to be resolved.

Valuations of the *sustainable uses* of habitat have been carried out, eg for medicinal plants and non-timber forest products. Hence there is considerable scope for at least securing minimum values for biological diversity through the use of approaches focused on market values.

A CLASSIFICATION OF VALUATION PROCEDURES

There are basically two broad approaches to valuation, each comprising a number of techniques:

1 direct approaches; and

2 indirect approaches.

The direct approach looks at techniques which attempt to elicit preferences directly by the use of survey and experimental techniques, such as the contingent valuation and contingent ranking methods. People are asked directly to state or reveal their strength of preference for a proposed change. In contrast, indirect approaches are those techniques which seek to elicit preferences from actual, observed market based information. Preferences for the environmental good are revealed indirectly when an individual purchases a marketed good to which the environmental good is related in some way.

THE DIRECT VALUATION APPROACH

In the direct approach, an attempt is made to elicit preferences by either experiments or questionnaires.

Experiments

If an analyst wanted to know how much people value a potential new recreation site, the recreation site could be created and an entrance fee charged. The analyst then observes how many people actually use the site, in effect exchanging money for the recreation and aesthetic experience of visiting the site. Alternatively, if the analyst wanted to know how much people would be willing to pay to live in a city with improved water quality, water quality standards and property taxes would be raised in some cities and not in others. The analyst could then see how many people found it worthwhile to move to cities with improved water quality and higher taxes. In practice, large scale experiments of this type are very difficult to design and implement, although small scale experiments have been carried out successfully.

Questionnaires (surveys)

There are two types of questioning that can be undertaken:

1 *Eliciting Rankings* — this is similar to contingent valuation (see below) except that the questioner is content to obtain a ranking of preferences which can later be `anchored' by the analyst in a real price of something observed in the market. This is known as the *Contingent Ranking Method* (CRM).

2 *Eliciting Values* — people are asked directly to state or reveal `what they are willing to pay (WTP) for some change in provision of a good or service or to prevent a change' and/or `what they are willing to accept (WTA) to forgo a change or tolerate the change'. A contingent market encompasses 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 and respondents are assumed to behave in an identical way to that 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. This is known as the *Contingent Valuation Method* (CVM), and measures precisely what the analyst wants to know — the individual's strength of preferences for the proposed change — and can be used not only for non-marketed goods and services, but for market goods as well. If people were able to understand clearly the change in environmental quality being offered, and answered truthfully, this direct approach would be ideal. However the central problem with the approach is whether the intentions people indicate *ex-ante* (before the change) will accurately describe their behaviour *ex-post* (after the change) when they face no penalty or cost associated with a discrepancy between the two. This is known as `strategic bias' and occurs if there is a `free rider' problem.

The contingent valuation method

Interest in CVM has increased over the last decade or so because, firstly, it will be the only means available for valuing non-use values — the values obtained from indirect techniques are not aimed at capturing non-use values. Secondly, estimates obtained from 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 as scientific sampling theory, benefit estimation theory, computerized data management and public opinion polling have improved.

There are three basic parts to most CV survey instruments:

1 A hypothetical description (scenario) of the terms under which the good or service is to be offered is presented to the respondent. This will include information on when the service will be available, how the respondent will be expected to pay for it, how much others will be expected to pay, what institutions will be responsible for delivery of the service, the quality and reliability of the service.

2 The respondent is asked questions to determine how much he would value a good or service if confronted with the opportunity to obtain it under the specified terms and conditions. These questions take the form of asking how much an individual is WTP or WTA for some change in provision. Depending on the preferred elicitation format, econometric models are then used to infer a WTP for the change. An aggregate welfare measure can be calculated by multiplying a favoured measure of response central tendency (mean or median) over a relevant population of users.

3 Response validity is tested by relating WTP responses to respondent socioeconomic and demographic characteristics. Confirmation of *a priori* expectations of the relationship between WTP income, age and other variables is a good indication of meaningful responses.

CVM: elicitation, bias and acceptability

Respondents can indicate their choice or preference in a number of ways. One is to answer questions relating to whether or not they would want to purchase the service if it cost a specified amount. These are known as discrete or *dichotomous* choice questions. Another possibility is to ask respondents direct questions about the most they would be willing to pay for the good or service — known as continuous or open-ended questions. These two types can be combined in a CV questionnaire to create different ways of eliciting the valuation information, eg a bidding game. In addition, respondents may be shown a list of possible answers in the form of a `payment' card, and asked to indicate their choice, although this requires a careful determination of the range of possible answers. The appropriate choice for a specific problem is a matter of judgement on the part of the analyst. Many recent studies have opted for the dichotomous choice format for its bias-reducing properties. In other words a `take it or leave it' question approximates a market transaction familiar to most respondents and minimizes the incentive to give a strategic response (see below). Choice of format also matters because the determination of mean or median WTP measures from yes-no data requires a higher degree of statistical competence. In essence, the required analysis uses qualitative choice models to calculate an expected mean WTP value (see Loomis, 1988). There is no one correct procedure for this analysis, and several of the assumptions routinely employed are the subject of current debate among practitioners (Hanemann, 1984, 1989; Johansson et al, 1989).

An assessment of the technical acceptability of CVM involves looking at various methodological issues, which we divide into issues of reliability, bias and validity.

Reliability: This looks at the degree to which the variance of WTP responses are attributable 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 value. The variance arises as a consequence of true random error (essential to the statistical process); sampling procedure (variance is minimized by ensuring a statistically significant sample size); the questionnaire/interview itself (it is important for reliability to ensure that the CVM scenario is as realistic and familiar to the respondent as possible). In order to assess reliability, a number of practitioners have advocated the use of replicability tests, ie repeating an experiment using different samples to see if there is correlation between the variables collected. Though few such tests have been carried out in practice due to their expense, Heberlein (1986); Loehman and De (1982) and Loomis (1989, 1990) have undertaken such testing and found significant correlation between WTP in the test and retest.

Bias: A number of different types of bias can be identified:

1 *Strategic bias* — the problem of strategic bias has long worried economists. The behaviour necessary for this kind of bias depends on the respondent's perceived payment obligation and his expectation about the provision of a good. Where individuals actually have to pay their reported WTP values then there is the temptation to understate their true preferences in the hope of a free ride. Or, if the price to be charged for the good is not tied to an individual's WTP response, but the provision of the good is, then over-reporting of WTP may occur in order to ensure provision. Empirical investigations of strategic bias are well documented. One approach to testing for strategic bias argues that if true WTP bids are theoretically normally distributed, strategic behaviour would bias this distribution towards zero (Brookshire et al, 1976). However this test has been criticized on the grounds that bimodal distributions can be posited on the income characteristics of the respondent population. Minimization of occurrence of strategic behaviour can be achieved by framing the CVM questions in an incentive compatible way such that this type of behaviour is not induced. One particular approach is to ask respondents to make bids for a good under three scenarios — only the highest bidders get the good; everyone gets the good if WTP is above a certain level; everyone with a positive WTP gets the good. The first scenario is assumed to give true WTP, the second has a weak free-riding incentive and the third a strong one. Empirical evidence suggests that the two latter scenarios do indeed produce WTP values below their true level. Such findings tend to come from open-ended format questions rather than discrete response approaches, where free-riding behaviour is likely to be minimized. Some authors suggest implementation of a property rights approach, in which respondents receive provision of a good relative to their given WTP in order to remove free riding. This is not applicable for most environmental public goods for which non-use and altruistic values act as a disincentive to free ride anyway. Overall, strategic bias problems have not been found to be a significant problem in practice.

2 *Hypothetical bias* — the hypothetical nature of the market in CV studies can render respondents' answers meaningless if their declared intentions cannot be taken as accurate guides of their actual behaviour. Some writers have looked at hypothetical bias in terms of increased bid variance and low model reliability, whereas others view the use of hypothetical markets as having other distinct problems. Research into hypothetical markets and their predictive ability has looked at the attitude-behaviour relationship, and experiments examining substitution of real for hypothetical markets.

The Fishbein-Azjen attitude behaviour model (1975) looks at the links between stated attitudes and actual behaviour. In order to minimize hypothetical bias, this model argues that the specified attitude (WTP scenario) must closely correspond to the specified behaviour (the precise good measured). Secondly, predictive power will be greater, the fewer the influencing relationships between a component in the model and behaviour. Thirdly, where a respondent is dealing with familiar behavioural situations, then attitude will be a better predictor of behaviour.

A survey of experimental tests reveals that by using a WTP format instead of a WTA format, hypothetical bias, which may be a significant problem in WTA studies, can be reduced to an insignificant level. The tests usually compare the hypothetical bids with bids obtained in simulated markets where real money transactions have taken place. Results from such studies suggest that the divergence between actual and hypothetical WTP is much less than that for WTA, the reason being that respondents are more familiar with payment rather than compensation scenarios (Hanley, 1990).

3 *The embedding problem* — there is evidence to suggest that people have problems understanding certain kinds of questions that depend on insights into their own feelings or their memory of events or feelings. This kind of problem will be very apparent in environmental issues because they evoke deeply held moral, philosophical and religious beliefs. One particular problem in this vein much looked at, is that respondents may interpret the hypothetical offers of a specific good or service to be indicative of an offer for a broader set of similar goods and services. This is known as the embedding problem since the value of the good being sought is embedded in the value of the more encompassing set of goods or services reported by the respondent. This problem is indicative of an even broader problem with obtaining accurate answers. For a single

individual the total amount they are WTP for improved environmental goods and services may be determined by the composition or components of the total set of environmental projects and policies to be funded. However this information is unlikely to be obtained from the aggregation of values based on a set of CV studies designed to measure individuals' preferences for narrowly defined environmental goods. Kahneman and Knetsch (1992) provide recent empirical evidence of embedding in a study looking at WTP for maintaining the quality of fishing lakes. Individuals were instead thought to purchase moral satisfaction in their WTP responses. This study was criticized on the grounds of having a poor instrument design — using telephone surveys, poor information — a single sentence to describe the good, and problems with the actual question framing rather than the underlying theory, which resulted in embedding. Problems remain with embedding. Willis and Garrod (1991) set the problem in the context of the theory of two-stage budgeting, where total income is firstly allocated to broad expenditure categories and then, secondly, subdivided within categories among specific items. Several mental accounts are thus set up, each referring to a category. Thus in responding to questionnaires, individuals may not consider the limits to, and other demands upon, their relevant mental account. This omission lies at the heart of the problem and is addressed by asking respondents to calculate total yearly budget for all environmental issues. Willis and Garrod (1991) test this for the Yorkshire Dales National Park and find no significant evidence of the problem. This result is supported by other similar studies. A variant of the problem is where WTP for a category group of goods is less than the sum of WTP responses regarding the specific good contents of that category, ie where WTP is asked in the context of other goods then this amount may tend to be less than if asked in isolation of other goods. Empirical evidence is mixed. Some studies have extended this contextual problem and found an ordering effect such that the higher a good is up a list of goods to be valued, the higher the WTP response.

4 *Information bias* — the quality of information given in a hypothetical market scenario almost certainly affects the responses received. Empirical evidence suggests only a weak information bias with some studies finding a threshold effect for information build up, below which no bias is detectable but above which a positive and weak effect is found. Other studies have found no significant information bias, though bid variance was found to fall as information increased. A number of writers have argued that information will always affect WTP but that this result applies to all goods, be they public or private. In the context of unknown or lesser order species, the issue of information provision as the basis of a valuation response is clearly vital. If CV use is to be extended beyond celebrated species and ecosystems, the effects of information provision must be addressed. Few experimental studies have investigated the extent of an information threshold in the context of biodiversity values. One current debate among practitioners is, however, jammed on an analogous issue of an information constraint on the validity of existence value statements (NOAA, 1994). More specifically, should uninformed respondents be informed, and how does this process affect their eventual responses to CV questions? If there is no information provision, should the responses of uninformed respondents count, and what does this imply for the range of subjects suitable for CV studies?

5 *Aggregation bias* — there may be problems in aggregating individual valuation responses. Analysts will often wish to summarize respondents' answers to valuation questions in terms of the mean willingness to pay for the good or service, or develop an aggregate benefit estimate for a community or region. Two types of problems here are sampling errors and insufficient sample size. Sampling errors include a non-random sample being selected and used. This may result from non-responses to the questions. Non-responses are more likely to occur for certain types of individuals who are not randomly distributed in the population. If the size of the sample is small, there is a risk that the characteristics of the sample will not be representative of the general population, thus resulting in findings which suffer wide confidence intervals. Furthermore, non-normal WTP distributions can cause the sample mean to be biased by the major tail of the distribution, necessitating the use of truncated means as an aggregate measure of welfare.

Often, on-site surveys will ignore the non-use values held by non-visitors such that additional random sample off-site surveys will be needed to estimate non-use values.

Empirical studies have found total non-use value is significant and can even exceed total use value.

6 *Interviewer and respondent bias* — the way interviewers conduct themselves and interviews can influence responses. Bias can be minimized by using mail or telephone surveys though this can mean less information is forthcoming and give rise to hypothetical bias. Mail surveys also give low response rates. Another variant of this problem is compliance bias, whereby the respondent tries to guess the `correct' answer or not give the questions proper considerations. To minimize this problem professional interviewers should be used, and they should follow the wording of the questionnaire *exactly*, with the respondents being offered a choice of prepared responses.

7 *Payment vehicle bias* — a number of studies have found that WTP varies depending on whether an income tax increase or an entrance fee is used as a payment vehicle (method of payment for the good). To minimize this bias, controversial payment vehicles should be avoided and a method used which is most likely to be used in real life to elicit payment for the good in question.

8 *Starting point, anchoring and discrete bid level bias* — The suggestion of an initial starting point in a bidding game can significantly influence the final bid, eg choosing a low (high) starting point leads to a low (high) mean WTP. The use of starting points can reduce the amount of non-responses and the variance in open-ended type questionnaires, though it also may result in respondents not giving their answer serious thought and taking a cognitive short cut in arriving at their decision. One solution to this problem is the use of what is called a `payment card' whereby the respondent selects a bid from a range shown on the card. However this can result in an `anchoring' of bids within the range of bids asked. Optimal bids should be set such that the lowest bid results in all respondents accepting it, and the highest bid results in all respondents rejecting it. Within this range, bid levels should reflect the distribution of bids so that, optimally, each bid interval reflects the same proportion of the population. A recent study by Bateman et al (1992), used a large sample open-ended format WTP question in order to estimate the distribution and range of WTP bids. A bid function was then estimated such that a probability of discrete bid acceptance curve could be mapped out. Eight WTP bid levels were subsequently chosen for a dichotomous choice experiment. The results were compared with an open-ended experiment and it was found that a dichotomous choice respondent was more likely to assent to the question `are you WTP £x?' than an open-ended respondent is likely to state a WTP of £x or above. Several factors may have influenced this result. Open-ended format studies are subject to free rider problems whereas dichotomous choice is not; dichotomous choice formats may be subject to interviewer bias and are more likely to exhibit anchoring bias, thereby biasing upwards the mean WTP. To conclude, open-ended approaches are likely to provide a lower bound WTP estimate below which true WTP is unlikely to lie, while dichotomous choice approaches provide an upper bound WTP estimate above which true WTP is unlikely to lie.

Validity: There are three categories of validity testing used in CVM studies:

1 *Content validity* — this looks at whether the WTP measure estimated in a CV study accurately corresponds to the object being looked at (the construct). Such testing cannot be formalized, resulting in analysts having to decide in a subjective manner whether a CVM has asked the correct questions appropriately, and if the WTP measure is in fact what respondents would actually pay for a public good if a market existed. Five `Reference Operating Conditions' have been proposed by Cummings et al (1986) for enhancing the validity of CV studies. It is thought that due to improvements in survey questionnaire design, content validity is not as great a problem as first thought.

2 *Criterion Validity* — here the CVM estimates are compared with the `true' value (the criterion) of the good in question. This is not feasible for many environmental goods (and is why CVM is carried out in the first place). However experiments comparing hypothetical WTP sums from CVM with `true' WTP, as determined by simulated markets using real money payments, have been carried out as mentioned in the earlier section on hypothetical bias. These find that in general, WTP format CVM studies give valid estimates of true WTP, though this is not the case for WTA.

3 *Construct validity* (including convergent and theoretical validity) — theoretical validity tests whether the CVM measure conforms to theoretical expectations, and convergent validity tests whether the CVM measure is closely correlated with measures of the good found using other valuation techniques.

Theoretical validity tests have centred on examining bid curve functions to see if they conform to theoretical expectations, eg whether elasticities are correctly signed and feasibly sized; tests on the significance of explanatory variables (by looking at simple `t' statistic tests, and the explanatory power of bid functions).

Convergent validity compares CVM measures with revealed preference techniques such as travel cost and hedonic pricing (see later). However the methods compared are usually measuring different theoretical constructs, eg, CVM measures use and non-use values whereas travel cost only measures use values. Furthermore, CVM provides *ex-ante* measures of WTP whilst hedonic pricing and travel cost estimates are from ex-post contexts. As such, the usefulness of convergent validity testing is not as great as at first thought.

Analysis of WTP responses

There are three ways in which CVM information is typically analysed:

1 Analysts examine the frequency distribution of the responses to the valuation questions.

2 Cross tabulations between WTP and socioeconomic variables, etc, are looked at.

3 Multivariate statistical techniques are used to estimate a valuation function that relates the respondent's answer to the socioeconomic characteristics of the respondent.

These analyses are used to see if respondents' answers are consistent with theory and to establish statistical relationships that can be used in the aggregation of sample responses to the overall population under study. Before the analyses can be undertaken, the data must be `cleaned' by removing `protest responses' of individuals who reject the hypothetical scenario and refuse to give meaningful answers. This can be done by setting an upper limit on how much a respondent's bid could be above the mean bid of the sample, or by seeing if respondents bidding very high or low have the socioeconomic characteristics that one would expect to be associated with such a response. Obviously such judgements are subjective and require careful consideration of field conditions, questionnaire, and analysis of the data.

Analysis of frequency distributions of WTP responses: Answers to open-ended valuation questions yield a set of point estimates of WTP. Statistics such as the mean, median and frequency distributions can be found for data sets of point estimates of WTP. Data such as mean estimates of WTP can provide estimates of total value of the good or service, alternatively the frequency distribution can be used to estimate the percentage of the population that would choose to purchase the good if it were offered at a specific price. Answers for dichotomous choice questions can be summarized in a way that provides similar information to the frequency distribution of point estimates described above. The percentage of respondents that agreed to pay each specified price can be found and then the relationship between these percentages and the price of the good can be graphed.

Cross tabulations of WTP responses with socioeconomic characteristics of the respondent and attitudes towards the environment: When point estimates of WTP are available for respondents, the analyst can calculate the mean WTP bid for different groups of respondents, so as to address the question of who is willing to pay the most for the good and why. If these cross tabulations of WTP bids and socioeconomic or attitudinal information reveal the effects one would expect from demand theory then the analyst has greater confidence in the quality of the data and greater insight into the factors that determine an individual's WTP. Cross tabulations for dichotomous choice questions are also possible but require large sample sizes since there may otherwise not be enough independent observations or degrees of freedom to carry out tests of differences between groups that have much statistical power.

Multivariate analyses of the determinants of WTP responses: Here the approach is to estimate a valuation function that relates the hypothesized determinants with the WTP responses. The determinants typically used include socioeconomic and demographic characteristics of the household and prices and availability of substitute goods and services. Open-ended questions will give a continuous measure of WTP for the good or service such that Ordinary Least Squares (OLS) models

can be used to explain the variations in the dependent variable. OLS requires, of course, that the determinants of the WTP responses be exogenous in order for the parameter estimates to be unbiased and consistent. It is thought that if R^2 values for valuation functions fall below 0.15 then the credibility of the values should be called into question. For dichotomous choice questions the responses are discrete and so OLS is unsuitable for estimation of the valuation function. Instead a variety of discrete choice models are available to explain the probability that a respondent will give a *yes* response to the valuation question. Again, the same kind of independent variables as above are used to explain the respondents' answers. For computational ease, most studies use logit or probit models. Such discrete choice models can be used to derive estimates of the economic value of the good and also of the relationship between the percentage of respondents agreeing to pay and the price offered whilst controlling for socioeconomic characteristics of the respondents and other factors.

Conclusion on CVM

It is important to get accurate, reliable answers to CV questions. In a report to the US National Oceanic and Atmospheric Administration (NOAA) committee, Arrow et al (1993) have offered a set of guidelines that they believe CV researchers should follow in order to ensure that CV studies provide accurate, reliable information (see below). The best prospects for use of CVM are in attempting to find WTP for an environmental gain, and when familiar goods are being looked at such as local recreational amenities. WTP and WTA for environmental losses are more problematic. Despite the pedigree of the Nobel laureate panel, the guidelines are in no sense a last word in the CV debate. Persisting areas of disagreement range from reliability of small sample sizes to the issue of previous information. Similarly, few of the issues concerning the psychological and cognitive processes of response formulation have been definitively resolved (Willis, 1993; Schkade and Payne, 1994). Despite these reservations, CV remains a promising option for biodiversity valuation. As already noted, the technique is the only way to elicit non-use values directly. Secondly, the potential for information provision and exchange during the survey process or during respondent focus group exercises, or using verbal protocols (`think aloud analysis'), offers scope to experiment with respondent knowledge and understanding of biodiversity. Recent improvements on the CV design process, employing *stated preference methods* (SPM) offer similar promise (Adamowicz, 1994; Louviere, 1994). Developed in the marketing and transport fields, SPM elicit consumer responses to a broader range of subject resource attributes than normally employed in a CV survey. Lastly, even if specific CV surveys are deemed unreliable for use in cost-benefit appraisal or for legal purposes, they can at least be treated as a surrogate referendum for determining conservation priorities based on public preferences (Blamey and Common, 1993).

GUIDELINES FOR CONDUCTING CONTINGENT VALUATION STUDIES

(Adapted from the report of the National Oceanic and Atmospheric Administration Panel on the Contingent Valuation Method, Arrow et al, 1993)

GENERAL GUIDELINES

1 *Sample type and size* — probability sampling is essential. The choice of sample specific design and size is a difficult, technical question that requires the guidance of a professional sampling statistician.

2 *Minimize non-responses* — high non-response rates would make CV survey results unreliable.

3 *Personal interview* — it is unlikely that reliable estimates of values can be elicited with mail surveys. Face-to-face interviews are usually preferable, although telephone interviews have some advantages in terms of cost and centralized supervision.

4 *Pretesting for interviewer effects* — an important respect in which CV surveys differ from actual referendum is the presence of an interviewer (except in the case of mail surveys). It is possible that interviewers contribute to `social desirability' bias, since preserving the environment is widely viewed as something positive. In order to test this possibility, major CV studies should incorporate experiments that assess interviewer effects.

5 *Reporting* — every report of a CV study should make clear the definition of the population sampled, the sampling frame used, the sample size, the overall sample non-response rate and its components (eg, refusals), and item non-response on all important questions. The report should also reproduce the exact wording and sequence of the questionnaire and of

other communications to respondents (eg, advance letters). All data from the study should be archived and made available to interested parties.

6 *Careful pretesting of a CV questionnaire* — respondents in a CV survey are ordinarily presented with a good deal of new and often technical information, well beyond what is typical in most surveys. This requires very careful pilot work and pretesting, plus evidence from the final survey that respondents understood and accepted the description of the good or service offered and the questioning reasonably well.

GUIDELINES FOR VALUE ELICITATION SURVEYS

7 *Conservative design* — When aspects of the survey design and the analysis of the responses are ambiguous, the option that tends to underestimate willingness to pay is generally preferred. A Conservative design increases the reliability of the estimate by eliminating extreme responses that can enlarge estimated values wildly and implausibly.

8 *Elicitation format* — the willingness-to-pay format should be used instead of compensation required because the former is the conservative choice.

9 *Referendum format* — the valuation question generally should be posed as a vote on a referendum.

10 *Accurate description of the programme or policy* — adequate information must be provided to respondents about the environmental programme that is offered.

11 *Pretesting of photographs* — the effects of photographs on subjects must be carefully explored.

12 *Reminder of substitute commodities* — respondents must be reminded of substitute commodities. This reminder should be introduced forcefully and directly prior to the main valuation to assure that the respondents have the alternatives clearly in mind.

13 *Temporal averaging* — time dependent measurement noise should be reduced by averaging across independently drawn samples taken at different points in time. A clear and substantial time trend in the responses would cast doubt on the `reliability' of the value information obtained from a CV survey.

14 *`No-answer' option* — a `no-answer' option should be explicitly allowed in the addition to the `yes' and `no' vote options on the main valuation (referendum) question. Respondents who choose the `no-answer' option should be asked to explain their choice.

15 *yes/no follow-ups* — yes and no responses should be followed up by the open-ended question: `Why did you vote yes/no?'

16 *Cross-tabulations* — the survey should include a variety of other questions that help interpret the responses to the primary valuation question. The final report should include summaries of willingness to pay broken down by these categories (eg, income, education, attitudes toward the environment).

17 *Checks on understanding and acceptance* — the survey instrument should not be so complex that it poses tasks that are beyond the ability or interest level of many participants.

THE INDIRECT VALUATION APPROACH

Indirect approaches are those techniques which seek to elicit preferences from actual, observed market based information. Preferences for the environmental good are revealed indirectly, when an individual purchases a marketed good which the environmental good is related to in some way. The techniques included here are as follows:

- hedonic price and wage techniques;
- the travel cost method;
- avertive behaviour; and
- dose-response and replacement cost techniques.

They are all indirect because they do not rely on people's direct answers to questions about how much they would be willing to pay (or accept) for an environmental quality change. As such, the

emphasis of these techniques is mainly on their contribution to valuing *biological resources* as opposed to biodiversity *per se* (see Chapter 1). That is, they allow inference of values for, say, national parks or forest reserves, but the absence of direct enquiry restricts the extent of subject goods. Obtained values will be sufficient for cost-benefit purposes, but will rarely reflect less appropriable biological wealth. As such, the techniques provide only a lower bound estimate of the value of a particular biological resource.

The indirect group of techniques can be divided into two categories: surrogate market approaches and conventional market approaches.

Surrogate markets

Surrogate market techniques involve looking at markets for private goods and services which are related to the environmental commodities of concern. The goods or services bought and sold in these surrogate markets will often have as complements (or attributes) and substitutes the environmental commodities in question. Individuals reveal their preferences for both the private marketed good and the environmental good when purchasing the private good. They leave what is called a `behavioural trail' as they make actual decisions that affect their lives. These techniques are therefore sometimes preferred by policy makers because they rely on actual choices rather than the hypothetical choices involved in the direct approaches. Surrogate market approaches include hedonic techniques and household production function techniques. The latter include the travel cost method which has been widely employed to detect the behavioural trail for biological resource preferences.

Household production functions

The household production function (HPF) approach argues that the environmental resource and private goods are demanded together with time, as intermediaries in a household's service flow or `well-being' production process. The approach places values on environmental resources by specifying some familiar structural relations (restrictions) between the environmental services of interest and other private goods. In the HPF approach, expenditures on commodities that are *substitutes* or *complements* for the environmental characteristic are used to value changes in that environmental characteristic. Thus, noise insulation is a substitute for a reduction in noise at source; travel is a complement to the recreational experience at the recreation site (it is necessary to travel to experience the recreational benefit). The values of the environmental resource are found by looking at changes in the expenditure on goods that are substitutes or complements to the environmental resource.

The *travel cost approach* uses observed expenditures on the travel to recreational sites to estimate the benefit arising from the recreational experience. This approach has relevance for valuing ecotourism. It can also be used to value the benefits of forest and woodland conservation for fuelwood (using travel time as a measure of the value of the fuelwood), and similarly for water supply (using travel time as a proxy for the value of improved water supply facilities). The approach typically uses information on money and time spent by people in getting to a site to estimate willingness to pay for a site's facilities or characteristics. The problem here is that many recreation sites charge a zero or negligible price which means that it is not possible to estimate demand in the usual way. However, by looking at how different people respond to differences in money travel cost (including transport, admission and the value of time, etc) we can infer how they might respond to changes in entry price, since one acts as a surrogate price for the other and variation in these prices results in variation in consumption.

The travel cost demand function is interpreted as the derived demand for a site's services and depends on the ability of a site to provide the recreation activity. Only use values are therefore considered, with existence and option values being ignored. Since the recreation activity takes place at specific sites that have observable characteristics and measurable travel costs, then recreational service flows are described as site specific. The approach can therefore provide us with estimates of the value of the site itself and, by observing how visitation rates to a site change as the environmental quality of the site changes, provide us with values for environmental quality itself.

The travel cost approach makes the central assumption that visit costs can be taken as an indication of recreational value. However, if individuals have changed their place of residency so as to be close to a site then the price of a trip becomes endogenous and the central assumption is violated. The estimated demand curve will lie below the true demand curve and so consumer surplus will be underestimated. A similar challenge to the central assumption also arises in cases where the on-site

time is not the only objective of the trip, eg where multi-purpose trips are made. The problem of multi-purpose trips presents particular problems for using the travel cost method to value internationally renowned sites which are typically visited as part of a wider experience. Several methods for allocating trip costs over all sites visited have been suggested (Mendelsohn et al, 1992), although none is considered ideal.

Data requirements: The data requirements of the approach are fairly substantial. A survey must be carried out to establish the number of visitors to a site, their place of origin, socioeconomic characteristics, the duration of the journey and time spent at the site, direct travel expenses, values placed on time by the respondent (see later), purpose of the visit other than visiting the site (multipurpose visits raise problems for the technique), and a whole range of environmental quality attributes for the site and substitute sites (see the earlier discussion on environmental quality measures). All of this data collection is expensive and time consuming to carry out.

The socioeconomic characteristics will include things like income, age, a measure of education, sex, race, and perhaps some measure of the subjective strength of preference for the particular type of recreation being offered.

Time costs: Since the cost of visiting a site consists of the transportation costs plus the costs of the time taken to get to the site and the time spent at the site, the role of time is critical to the estimation of travel costs. Time costs are included because time has an opportunity cost, for example, one could work instead. We need to know what elements of time are to be included in the travel costs, what money values to use for these time costs, and how their inclusion will affect the demand and benefit estimates.

If time costs are ignored then benefits and demand will be biased, since, for example, two visitors to a site may have had to travel different distances to the site whilst having equal money travel costs but requiring substantially different times to get to the site. Unless time costs are included, visitation rates may appear to be equal for the two zones and willingness to pay for the site will be equal. The effects of both time costs and money travel costs on visitation rates therefore need to be estimated separately, but since the two may be highly correlated and so separate estimation difficult, time costs are given a money value using some shadow price of time and are lumped together with the transportation costs. Time at the site should also be included in travel costs because it may not be independent of the distance travelled. The shadow price of time at the site and time getting to the site may however be different. Any difference will be due to individuals deriving pleasure from the journey to the site, eg, by taking a scenic route. If no pleasure or displeasure is forthcoming then the shadow prices are the same.

The marginal wage rate is often used as an appropriate shadow price of time, since this reflects the opportunity cost of time between working and not working. However this trade-off may be distorted by institutional constraints such as maximum working hours, taxation etc; or, using the wage rate may be inappropriate for certain groups such as the unemployed. Previous empirical work has suggested that the shadow price of time may be substantially less than the wage rate and lie somewhere between 25 and 50 per cent of the wage rate with a value of 33 per cent of the wage rate being appropriate (Cesario, 1976). Some studies determine the proportion of the wage rate to use within the estimation procedure, eg Common (1973), McConnell and Strand (1981).

Exclusion of time costs in general will result in a more elastic (flatter) demand curve and bias the benefit estimates downwards. Exclusion of on-site time costs, if they are not independent of distance travelled and vary inversely with it, will result in a less elastic demand curve and an overestimate of benefits.

Specification and estimation issues: A trip-generating function estimating the number of visits to a site as a function of travel costs and the socioeconomic variables is the first step in specifying a demand relationship. Specification of the functional form is crucial to the benefit estimates obtained. Standard statistical techniques will in general not be able to discriminate in favour of one specification or another. In practice the choice of functional form needs to be determined empirically on an individual study basis. However a number of studies have found that the visitation rate equation is best estimated using a semi-log form, ie, the logarithm of the number of visits to the site is regressed against travel cost, etc. Generally, it has been found that log-linear and semi-log specifications increase valuations relative to results found using a semi-log for the explanatory variables (Smith and Kaoru, 1990).

The second stage in specifying the demand relationship involves explaining the variation in visitation rates across sites according to site characteristics. One procedure for doing this is to use the two stage varying parameter model of Smith and Desvousges (1986). Here, a trip-generating function is estimated separately for each site, without including any environmental quality variables as above. The second stage is then to explain the difference in the coefficients on the travel cost terms by regressing them on the environmental quality variables. The coefficient on the quality variable then shows how the demand curve shifts as quality of the site changes and thus can be used to estimate the benefits of a change. Using this second stage procedure also reduces the risk of multicollinearity problems especially where sites possess multiple environmental attributes which may be highly correlated. Inclusion of such attributes as separate variables in a single stage estimation will lead to multicollinearity.

There are a number of major statistical estimation problems with the travel cost approach:

1 Misspecification of the functional form can lead to biased parameter estimates.

2 The number of visits to a site can only be a non-negative variable such that continuous estimation techniques such as OLS are inappropriate. Discrete choice models of behaviour such as the multinomial logit model should therefore be used. This looks at the probability that a particular site will be visited, depending on the attributes of that site and other sites, and on the households' characteristics. Since individuals will make no visits to some of the sites then there will be some zero values for the visitation rate variable. Using OLS therefore implies that a change in the quality of a site will have an effect on visitation rates, even if the site is not visited. Clearly this is incorrect and so the logit model is used instead. In this, the benefit per visit of an improvement in site quality can be estimated from the logit equation if a measure of travel cost is included. An increase in quality will increase the probability of visiting a site. The benefit per visit is then found by calculating the compensating increase in travel cost that would leave the probability of visiting the site unchanged. This requires total differentiation of the logit equation. Use of such models also removes the problem of substitute sites, where individuals are faced with the choice of many sites at various distances and with different quality characteristics. Exclusion of the impact of substitute sites on demand will lead to biased estimates.

3 In any data set we will have information on people who actually have visited the site, but not on non-participants. Non-users need to be included to see what determines participation. This problem is known as *truncation bias* and has been found to have a significant impact on parameter estimates such that the estimated demand curve is flatter than the true one. One suggestion has been to use *maximum likelihood estimation* instead of OLS in order to counter this problem. However the evidence here is mixed (see Smith and Desvousges, 1986; Kling, 1987, 1988; and Smith, 1988). If there are systematic influences on the participation decision then a sample selection problem exists.

In conclusion, we can say 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 reservations as to its use, particularly concerning the large amounts of data required which are 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.

Whereas the travel cost method employed complementarity between market and environmental goods, the second HPF approach, the *averting behaviour technique,* makes a substitutability assumption. Examples of averting behaviour include: looking at expenditures on improved ventilation in order to reduce the exposure to radon in houses; valuing the costs of siltation from upstream erosion by looking at the expenses that farmers incur when installing protection structures; valuing health hazards from river water by looking at WTP for bottled water, filtration devices and private well installation.

To undertake such an estimation, data on the environmental change and its associated substitution effects are required. Fairly crude approximations can be found simply by looking at the change in expenditure on the substitute good arising as a result of some change in the environmental

commodity of interest. Alternatively, if the marginal rate of substitution between the environmental commodity and the private good, which can be found from known or observed technical consumption data, is multiplied by the price of the substitute, then the value per unit change of the environmental good can be found.

Strictly speaking, in order to apply this approach the averting behaviour must be between two perfect substitutes, otherwise an underestimation of the benefits of the environmental good will occur. Averting behaviours are never likely to involve perfect substitutes and even when they do, bias in the estimation of benefits can still occur. For example, if there is an increase in environmental quality, the benefit of this change is given by the reduction in spending on the substitute market good required to keep the individual on their original level of welfare. However when the quality change takes place the individual will not reduce spending so as to stay on the original welfare level. There will have been an income effect as well as a substitution effect between environmental quality and the substitute good. Expenditure will therefore be reallocated among all goods with a positive income elasticity of demand and so the reduction in spending on the substitute for environmental quality will not capture all of the benefits of the increase in quality. Further problems with the approach include the fact that individuals may undertake more than one form of averting behaviour to any one environmental change, and, that the averting behaviour may prevent the adverse effects of reducing the environmental good but may also have other beneficial effects which are not considered explicitly, eg, sound insulation may also reduce heat loss from a home. Furthermore, averting behaviour is often not a continuous decision but rather a discrete one — a smoke alarm is either purchased or not, etc. In this case the technique will again give an underestimate of benefits unless discrete choice models for averting behaviour are used.

Thus, simple avertive behaviour models, although having relatively modest data requirements, can give incorrect estimates if they fail to incorporate the technical and behavioural alternatives to individuals' responses to quality changes. Although the technique has rarely been used, it is a potentially important source of valuation estimates since it gives theoretically correct estimates which are gained from actual expenditures and thus have high criterion validity.

Hedonic pricing

The hedonic pricing method (HPM) is similar to the household production function approach since both make a complementarity assumption. With HPM an attempt is made to estimate an *implicit price* for environmental attributes by looking at real markets in which those characteristics are effectively traded. Thus, `clean air' and `peace and quiet' are effectively traded in the property market since purchasers of houses and land do consider these environmental dimensions as characteristics of property. The attribute `risk' is traded in the labour market. High risk jobs may well have `risk premia' in the wages to compensate for the risk. The two HPM markets of most interest, therefore, are:

- *Hedonic house (land) prices* — for valuing air quality, noise, neighbourhood features (parks etc). Given that different locations of property assets will have different levels of environmental attributes and that these attributes affect the stream of benefits from the property, then the variation in attributes will result in differences in property values (since property values are related to the stream of benefits) Ostensibly, land values in developing countries may well reflect the presence of soil conservation measures, access to fuelwood etc — characteristics favourable to the conservation of biological resources and incidental diversity. Similarly, property prices in developed countries frequently reflect surrounding aesthetic benefits of landscape or woodland. The hedonic price approach looks for any systematic differences in property values between locations and tries to separate out the effect of environmental quality on these values.
- *Wage risk premia* — for valuing changes in morbidity and mortality arising from environmental (and safety) hazards. Since labour markets in developing countries are unlikely to function so as to capture risk aversion, this approach currently has limited relevance in the developing country context. Note, however, that valuations from *developed* economies derived from wage risk studies may be adjusted to provide approximations of statistical life values for developing countries and economies in transition (Pearce, 1986).

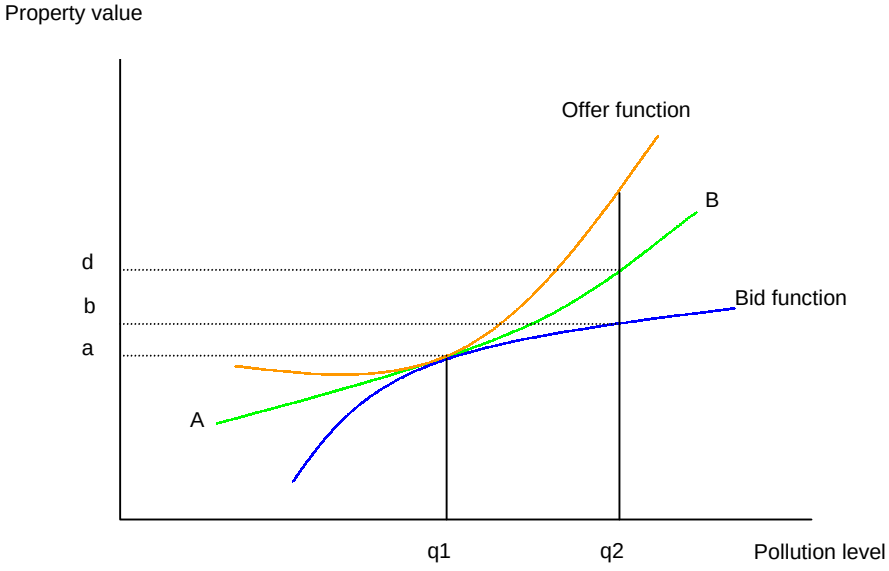
To find the demand function relating the quantity of the environmental attribute to individuals' WTP it is necessary to first define the market commodity (eg housing) and the environmental attribute of the market commodity (eg air quality). A functional relationship is then specified between the market price

and all the relevant attributes of the market commodity. This is called a *hedonic price function.* The hedonic price function is then estimated using multiple regression techniques from data on property values and the associated attributes of the property. We are thus able to find the hedonic price function coefficient on the attribute of interest (air quality) and this coefficient is known as the *marginal implicit price* of the attribute; it gives the additional amount of money that must be paid by an individual to buy an identical market good but with a higher level of the environmental attribute. Consider Figure 5.1. The analysis attempts to identify the slope of the curve AB, which shows the relationship between the level of air quality and the price of the property.

The curve AB in the diagram represents the result of a market equilibrium in which individuals buy property at some level of air quality and suppliers (owners or property developers) sell property with various air quality levels. The pollution level axis shows increasing levels of air quality. Individuals will buy property at some level of air quality according to their bid curves and suppliers supply property with this air quality level according to their offer curves, with points of tangency between the bid and offer curves giving equilibrium points on the hedonic price curve.

In benefit estimation we are interested in individuals' willingness to pay for better air quality. How does this relate to the following diagram? Well, say the level of air quality changes from q_1 to q_2, then individuals' willingness to pay for this change is given by the distance *ab* in the diagram. However, the estimated hedonic price function would tell us that the willingness to pay for the change would be the distance *ad,* and so gives an overestimate of the benefits of the change. To correct for this bias and so estimate the true inverse demand (willingness to pay) curve, a second stage to the procedure is used. This second stage of the analysis requires certain assumptions regarding the supply side of the market good in order to estimate the willingness to pay curve. This usually takes the form of a fixed supply assumption. A further statistical regression in which the marginal implicit price of air quality is regressed against the socioeconomic characteristics of individuals (including income) is undertaken to estimate the bid function.

Figure 5.1 *The hedonic price curve*



This second stage of the analysis is not always necessary. Freeman (1979) shows that if all individuals are identical in all respects such as income, preferences, etc, then the implicit price function over the range of air quality (the slope of the hedonic price function over the air quality range) will give the inverse demand function since the implicit price function shows the locus of points on individuals' inverse demand curves which will all coincide since individuals have the same preferences, income and so on. Although this assumption allows easy benefit estimation from just the hedonic price function it is obviously a fairly unrealistic case.

Alternatively we can make various assumptions regarding the shape of the inverse demand curve in order to estimate the benefits of the quality change. These are considered highly questionable and so we do not analyse them further here.

Data requirements: The data requirements of the approach are substantial. Data from a wide range of different properties are required with information on all features that influence the properties' value such as structural characteristics (number of rooms, size, etc), neighbourhood characteristics (`prestige', closeness to business and amenity areas, etc), and environmental characteristics (air quality, noise levels, etc), as on the property values themselves. In practice, cross sectional data tend to be used. This removes problems regarding changes in the economic well-being of an area which will tend to arise if time series data are used. Socio-economic data on individuals (such as income, age, education, etc) are also required if the second stage estimation procedure is carried out. Sufficient data of the variety to enable reliable estimation may be difficult to come by, especially in areas and countries containing a large amount of public sector housing. Moreover, in developing countries such data are likely to be incomplete or non-existent even in urban areas. This suggests that the technique has little potential for valuing biological resources in remote areas with undeveloped asset price markets.

The data on property values should come from actual market data but since only a small percentage of the total owner-occupied housing stock may be sold per year, then collection of a large enough sample of data may be difficult. Care must be taken to account for the effects of property taxation on property values, otherwise their use will result in an overestimation of benefits. A further problem is that property prices may be influenced by expected future changes in the property, and so the characteristics at the time of a sale may not adequately explain the selling price. Rental price data could be used to overcome this and are in any case the theoretically correct measure to use. However the rental market may be even less perfect than the property market in some countries. As an alternative Real Estate agent valuations could be used.

A major problem with hedonic price studies is that of multicollinearity — the fact that many of the explanatory variables will be related to one another, eg, sulphates and particulate measures will be collinear. This will result in difficulty in identifying which factor is determining movements in house prices. There are a number of ways to overcome multicollinearity. These include separating out the individual effects using statistical tests proposed by Klepper and Leamer (1984), or, formulating all data concerning individual pollutants into one proxy measure. Care must also be taken not to omit important explanatory variables as this will again bias the coefficient estimates.

Turning now to the data on environmental quality attributes, we need to know which pollutants are of interest and whether or not measures exist for them. Threshold levels may mean difficulty in measurement, or that pollution effects take a long time to show up. Temporal variations in concentrations may mean that it is best to use annual averages. We also face the same problems regarding objective and subjective measures of quality as we did for the travel cost approach. Subjective measures are what people's behaviour is based on and so are important for benefit estimation. Objective measures, however, are more extensively monitored for many pollutants. Will these objective measures coincide with people's perceptions? For measures such as suspended particulates which are readily perceptible and their effects apparent in terms of visibility, etc, there should be no problems. Sulphate pollution level measures are also thought to coincide with perceptions. When a single pollution variable is used, problems may arise with the pollution coefficient which may include the effects of omitted but collinear pollution variables.

Choice of functional form: The choice of functional form will have a significant impact on benefit estimates even if statistical tests find all the choices of form acceptable. In order for a preference to be made between functional forms, two questions should be asked about what properties the hedonic price function should possess. These are, whether the marginal implicit price of pollution is independent of the levels of the other attributes of housing and, whether the marginal implicit price depends on the pollution level itself, and if so, how? With regards to the first question only the log, and Box-Cox forms impose dependence on the levels of other attributes. The second question relates to the slope of the implicit price function and whether this is linear or not. In practice it is found that non-linear functional forms give better fits for the data, especially the log and semi-log variety. However, some studies use the Box-Cox transformation which allows the data to determine the precise form. Whichever functional form is used, care must be taken when transforming the estimated coefficients back to their original form (eg, from log back to antilog) since biases can result.

Other issues: The whole approach relies on the assumptions of a fixed supply of housing and a freely functioning and efficient property market. Individuals have perfect information and mobility such that they can buy the exact property and associated characteristics that they desire and so reveal their demand for environmental quality. In reality the housing market is unlikely to be so. As was mentioned earlier a large part of the housing stock may be in the public sector and so allocated subject to price controls. Furthermore market segmentation may exist whereby mobility between housing areas is restricted. To get around this problem separate hedonic price functions should be estimated for each segmented area. Mäler (1977) has criticized the fact that the technique implicitly assumes that households continually reevaluate their choice of location.

Another problem with the approach is that hedonic price includes the consumer valuation of not only present day benefits but also the stream of expected future (discounted) benefits from environmental quality, and as such will tend to overstate WTP.

Finally, the possibility that mitigating or averting behaviour by individuals may take place to avoid the effects of pollution, such as installing pollution filters, needs to be looked at (see the earlier section on averting behaviour). If this behaviour is unrelated to the characteristics of the property then it will reduce the value of the property and need not be measured separately. If changes do occur to the property then the value of the property will increase and so such changes need to be included in the hedonic equation.

To conclude, the hedonic approach is founded upon a sound theoretical base and is capable of producing valid estimates of benefits so long as individuals perceive environmental changes.

Both the HPM and HPF infer environmental prices from direct consumption of substitutes and complement goods. These prices are use values only and therefore understate total economic value of environmental goods. Data requirements for the HPF suggest that the technique is of limited value in the context of developing country biological resources.

CONVENTIONAL MARKET APPROACHES WHEN OUTPUT IS MEASURABLE

These approaches use market prices for the environmental service that is affected, or, if market prices are not an accurate guide to scarcity, then they may be adjusted by *shadow pricing.* Where environmental damage or improvement shows up in changes in the quantity or price of *marketed inputs* or outputs, the value of the change can be measured by changes in the total consumers' plus producers surplus. If the changes are small the monetary measure can be approximated by *market values.* Two approaches may be distinguished: the dose-response technique and the replacement cost approach.

The dose-response technique

This aims to establish a relationship between environmental damage (response) and some cause of the damage such as pollution (dose), such that a given level of pollution is associated with a change in output which is then valued at market, revealed/inferred, or shadow prices. Where individuals are unaware of the impact on utility of a change in environmental quality then direct WTP/WTA is an inappropriate measure and so dose-response procedures which do not rely on individuals' preferences can be used.

The technique is used extensively where dose-response relationships between some cause of damage such as pollution, and output/impacts are known. For example, it has been used to look at the effect of pollution on health, physical depreciation of material assets such as metal and buildings, aquatic ecosystems, vegetation and soil erosion. The approach is mainly applicable to environmental changes that have impacts on marketable goods and so it is unsuitable for valuing non-use benefits.

Damage actually done is found using a `dose-response function' which relates physical/biological changes in the ambient environment to the level of the cause of the change. The dose-response function is then multiplied by the unit `price' or value per unit of physical damage to give a `monetary damage function'.

The dose-response approach in its most basic form looks at environmental resources which lead to a marginal change in the output of a good sold on a competitive market and values the impact directly in terms of output changes valued at market prices. More formally, suppose the production function for a single output y is given by:

$$y = F(X, Z)$$

where X is a set of inputs and Z is the input of the unpriced environmental resource. Assume that we can measure the output y and that this output is sold on a market at prices. The price of inputs X is given by the price vector P. Now, if prices are not expected to change when supply of the environmental resource changes, then the economic value of the change in the supply of the resource is the value of the production change accompanying the change in resource availability at constant inputs of the other factors.

If the change in resource supply is large, but leaves prices unchanged, then the value of the resource supply change must be measured as the difference between the profit after the change and before the change, taking all changes in factor use into consideration. If, on the other hand, the dose *does* cause prices to change (working through quantity) then there are still changes in profit, but there will be an additional effect on consumers' surplus. In other words the total welfare effect of the dose becomes difficult to determine particularly if resource producers take their own avertive steps to mitigate pollution effects.

In practice all these effects need to be assessed within the framework of a market model. Modelling welfare effects can be an open-ended activity depending on the desired degree of sophistication. To the extent that a model misspecifies the dose-response relationship and related market effects, measurement of welfare change may be biased. Specification of an accurate dose-response relationship is a particularly difficult task requiring precise experimental data and consideration of any synergistic effects between pollutants impacting on the subject receptor. Moreover, the potential presence of damage thresholds and response function discontinuities in ecosystems, suggests that simple linear dose-response relationships may be inadequate.

To conclude, the dose-response approach is a technique that can be used where the physical and ecological relationships between pollution and output or impact are known. The approach cannot estimate non-use values. The approach is theoretically sound, with any uncertainty residing mainly in the errors of the dose-response relationship, eg, are there threshold levels before damage occurs, or discontinuities in the dose damage function? It is necessary to allow for the fact that the behaviour of individuals may change in response to changes in the environment. If this is not possible, but the direction of any bias resulting is known, then this should be stated. The approach may be costly to undertake if large databases need to be manipulated in order to establish the relationships. If the dose-response functions already exist though, the method can be very inexpensive, with low time demands and yet provide reasonable first approximations to the true economic value measures.

In the context of biological resources, application to fisheries, forestry and agriculture are particularly appropriate. In these sectors, causes of mortality and morbidity are relatively well understood, and reliable market or shadow price information is normally available. Identification of *in situ* dose response functions for a range of species and ecosystems represents a challenge which ecologists are beginning to address with reference to the fate of indicator species such as amphibians, mosses and lichens (Pechmann et al, 1991; Wyman, 1990). Subsequent translation into economic damages is, however, some way off.

The replacement cost technique

This technique looks at the cost of replacing or restoring a damaged asset to its original state and uses this cost as a measure of the benefit of restoration. The approach is widely used because it is easy to find estimates of such costs. The approach is correct where it is possible to argue that the remedial work must take place because of some other constraint such as a water quality standard. Under such a situation the costs of achieving that standard are a proxy for the benefits of reaching the standard, since society can be assumed as having sanctioned the cost by setting the standard. However, if the remedial cost is a measure of damage then the cost-benefit ratio of undertaking the remedial work will always be unitary. That is to say remedial costs are being used to measure remedial benefits. To say that the remedial work must be done implies that benefits exceed costs. Costs are then a lower bound of the true value of benefits. If, to pursue the water quality example, the standard has clearly been set without thought for costs, then using replacement costs as a measure of minimum benefits could be misleading.

Another situation where the replacement cost approach is valid would be where there is an overall constraint not to let environmental quality decline (sometimes called a `sustainability constraint'). In these circumstances replacement costs might be allowable as a first approximation of

benefits or damage. The socalled shadow project approach relies on such constraints. It argues that the cost of any project designed to restore an environment because of a sustainability constraint is then a minimum valuation of the damage done.

Information on replacement costs can be obtained from direct observation of actual spending on restoring damaged assets or from professional estimates of what it costs to restore the asset. It is assumed that the asset can be fully restored back to its original state. However some damage may not be fully perceived, or may arise in the long term, or may not be fully restorable. Benefits will therefore be underestimated. Another problem here is that restoration of damaged assets may have secondary benefits in addition to the benefits of restoration such that replacement costs will underestimate total benefits.

The replacement cost approach is useful for estimating flood protection and water regulatory services supplied by forested watersheds which provide natural barrages. Other possible applications include the replacement of traditional medicines and the costs of crop insurance to replace natural insurance afforded by genetically diverse traditional cropping systems.

Opportunity cost

In the opportunity cost approach no direct attempt is made to value benefits. Instead, the benefits of the activity causing environmental deterioration — say, a housing development — are estimated in order to set a benchmark for what the environmental benefits *would have to be* for the development *not* to be worthwhile. Clearly, this is not a valuation technique but, properly handled, it can be a powerful approach to a form of judgmental valuation. It is used here to indicate the kinds of economic returns that must be secured by biodiversity use if such land uses are to be economically preferred to the alternative land use.

CHOICE OF VALUATION TECHNIQUE

A growing body of case studies is indicative of the appeal of valuation techniques as additional tools to guide the conservation of biological resources. The value of resource diversity is a relatively new concern and some methods will be more easily tailored for the purpose of value elicitation.

All of the valuation techniques outlined have strengths and weaknesses as we have seen, and the decision on which valuation technique to use for a particular application requires experience and judgement on the part of an analyst. There are, however, some general points to consider when making a choice.

First, the technique should be technically acceptable with respect to its validity and reliability (see the section on the contingent valuation method). Measures obtained from the technique should be consistent and accurate. Methods suffering random errors require reliability checks to judge their predictive capacity. Methods suffering non-random error contain bias problems, thereby reducing reliability and the validity of the measurement results. Validity cannot be assessed solely on the basis of technique methodology but must be considered alongside practical predictive ability.

Reliability problems will occur if the sample size of the data is too small or a survey design is deficient. Reliability is closely related to bias which can vary depending on the good being looked at.

The hedonic pricing and travel cost approaches have weak validity since they assume the underlying theory is correct in order to generate results, whereas CVM can build in tests for reliability and validity. A more psychological approach can be taken with CVM, with direct psychometric testing of validity and reliability.

Second, the technique should be institutionally acceptable such that it fits into current decision making processes. There are differing views as to the acceptability of monetizing the environment.

Third, it is important to consider the needs of the user(s) of valuation studies who may prefer the use of one valuation technique over another. For example, estimates obtained from travel cost or hedonic property value models may be considered too theoretical or too complex. On the other hand it may be felt that contingent valuation estimates are too subjective and unreliable to support policy debate and discussion. The analyst carrying out policy work must be sensitive to such concerns. The technique should also be user friendly in terms of how easy or difficult it is to use in practice.

Fourthly, the financial cost of the study needs to be weighed against the value of the information gained.

Finally it should be remembered that it will often be possible to use more than one valuation technique and compare the results. The estimates of value obtained from all the methods described will be somewhat uncertain. If the analyst has multiple estimates, then they will have greater confidence in the magnitude of the value of the proposed change. Several of the valuation techniques typically use data from a household survey (eg, contingent valuation, travel cost model, and hedonic property value model). When the implementation of a valuation technique requires that primary data be collected with a household survey, it is often possible to design the survey to obtain the data necessary to undertake more than one valuation method.

6 VALUING BIODIVERSITY: ECONOMIC ESTIMATES

INTRODUCTION

Chapter 3 presented a decision rule for comparing the relative returns from the sustainable use of habitats with the opportunity cost. This decision rule was shown at two levels: that of the individual owner of the land, and that of the nation or the world as a whole. The former was used to explain how biodiversity loss can easily accompany everyday economic activity. The latter was used both to explain the kinds of values that are not captured by the individual when making land use decisions, and to suggest a rule for deciding how much biodiversity there should be. From that discussion it follows that if the divergence between national values and local private values is large, then the nation in question will have an interest in modifying market decisions to correct the relevant externalities. If the global values are additionally large, then there is yet a further reason to modify the private decisions of the landowner. In this case however it becomes important to determine whether these benefits accrue globally or disproportionately to the host or a limited group of states. In other words, which party has the greatest incentive to pay for the modification of landowners' decisions? What is being discussed is not the appropriation of the landowner's rights to his or her property, although that would be one extreme form of control. Rather it is the *provision of incentives designed to modify land use decisions.* The technical phrase is that we seek the *attenuation of property rights,* but preferably in such a way that the landowner in question is actually better off because of the control than without it. Instruments designed to serve these objectives such as tradeable development rights and land easements were suggested in Chapter 4. They are illustrative of the potential for mutual gain which underlies the philosophy of sustainable development — to have economic gains but not at the expense of the environment or, better still, to profit from environmental conservation.

Chapter 4 also showed that some biodiversity losses arise because of private gains that cannot be easily justified, eg the collection of subsidies for land clearance, intensive cultivation in the context of already existing overproduction, and so on. Here the issue is not one of making the landowner better off with the control: the original intervention itself has no real justification. Its removal will therefore be a cost to the private owner.

However, obeying an efficiency criterion for the sustainable use of land is not straightforward. Apart from the intrusion of government policy, the rule assumes that states and individuals can both recognize and realize a return from their resources. This in turn suggests a level of information which is often superior to that which exists in most countries. Indeed, if many population biologists are correct, many nations may never identify the majority of species and ecosystems they possess, still less attach indirect or non-use values to their use. In such circumstances, our decision rule is commonly reduced from a comparison of the total economic value to a simple trade-off of the direct use value versus the opportunity cost. *Biodiversity will be more prone to depletion when direct use values are not realized.*

Two issues therefore arise. First, how large are the unrealized benefits of conservation? Second, how can institutions be changed so that land users secure these unrealized benefits? This two-part approach defines the approach for the rest of the book, namely to demonstrate the economic values of conservation *relative to* the returns from land development; and to construct mechanisms for the *appropriation* of those values. The following section critically reviews the literature on how these benefits have been demonstrated using economic techniques. We then assess two of the more contentious arguments for resource preservation: the appropriation of value from plant genetic resources for medicinal purposes, and briefly, the value of genetic resources to agriculture. Direct use, option and quasi-option values associated with unscreened genetic material are often invoked to support area preservation. Yet the global value of these uses remains largely unquantified. Finally we review one of the only quantifiable manifestations of global existence value, the debt-for-nature swap.

A REVIEW OF ENVIRONMENTAL CONSERVATION VALUES

Numerous studies exist on the kinds of economic value that can be realized through conservation. A collection of such values is documented in Tables 6.1 to 6.4. The estimates vary in the degree of sophistication and the extent of benefit valuation. Non-use benefit estimation for example, is more frequently attempted in developed countries than in developing countries, where benefit estimation is usually restricted to use values. In addition there has been no systematic coverage of ecosystems or regions, although certain systems — particularly tropical forests and wetlands — are more common subjects of study. Few studies attempt to identify separate components of total economic value although many contingent valuation studies are designed to determine use and non-use values.

Differences in applied methodologies and a shortage of illustrative studies for many areas handicap any reliable generalization on the basis of existing data. Some ballpark figures are nevertheless beginning to emerge. Forest carbon sequestration and valuation has been extensively documented (Brown and Pearce, 1992). Similarly one review of 24 non-timber forest product valuation studies indicates a median value of about $50 per hectare per year, but with net values ranging between $1 and $420 (Godoy, Lubowski and Markandya, 1993). Clearly there is some scope for both harmonizing methodologies and deciding when a methodology is or is not suitably transferred between developed and developing countries. For the purpose of comparing estimates of similar species or ecosystem values, consistent data collection and analysis are vital for comparing like with like. Similarly, the compatibility of unmodified developed country methodologies for developing countries needs to be considered. Initial experience with contingent valuation and travel cost methods suggests that minimal modification is necessary for their application. Household surveys in rural areas, or sectors remote from the monetized economy present greater difficulties. One area for further research involves possible modification of economic techniques for use in conjunction with an established body of participatory and rapid rural appraisal methods. Contingent ranking exercises would, for example, be compatible with focus group or key informant interviews.

Table 6.1a *Conversion ratios for varying discount rates*

To convert r in row to r in column, multiply by cell value	r = 4	r = 5	r = 7	r = 10
r = 4, t = 10	1	0.95	0.87	0.76
t = 20		0.92	0.78	0.63
t = 30		0.88	0.72	0.44
r = 5, t = 10	1.05	1	0.91	0.80
t = 20	1.09		0.85	0.68
t = 30	1.12		0.80	0.61
r = 7, t = 10	1.15	1.10	1	0.87
t = 20	1.28	1.18		0.80
t = 30	1.39	1.24		0.76
r = 10, t = 10	1.32	1.26	1.14	1
t = 20	1.60	1.46	1.24	
t = 30	1.83	1.63	1.32	

Table 6.1 *Economic values — tropical forests*

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
Ecosystem type:	1 Sustainable harvesting in 1 hectare of Peruvian Amazon	3 Arising from sustained use of the Korup forest:	Lower bound option value may be inferred from the	2 Brazilian Amazon: ($199/pa)	1 Implicit ratios of 6.82, 2.14 or 2.3 depending on
Tropical Forests	(timber, fruit and latex $1987) Net present value hectare^{-1}	– Existence of watershed functions affording	current market value or foreign exchange earning	Direct use 15bn Indirect 46bn	alternative use, but subject to qualifications regarding
Sources: 1 Peters, Gentry and Mendelsohn (1989).	$6820 (local market values) relative to a net revenue $1000 h^{-1} from clear-felling which risks uncertain regeneration. $3184ha^{-1} plantations for timber and pulpwood or	protection to Nigerian and Cameroonian fisheries: NPV (£1989) £3.8m (approx $6.8m) or $54ha, assuming that the benefit starts to accrue in 2010 and beyond	potential of plant based pharmaceuticals. Attempts to gauge existence values in other contexts, rely on CVM to	Existence 30bn Total 91bn NPV (using Krutilla Fisher) $1296bn	local elasticity of demand for harvested forest products. Note that a similar exercise (12) in another area of Peruvian Amazon contradicts these
2 Gutierrez and Pearce (1992)	$2960ha^{-1} from cattle ranching. 2 Estimated contribution of direct use to Brazilian GNP	(2010 represents the time horizon by which the continued use of the forest resources — in the absence of	report WTP/WTA. To date only one study relating directly to tropical forests is available (10), although this	10 CVM survey of villagers willingness to accept compensation (WTA), to forgo use	estimates with a ratio of about 30 in favour of logging and rotation cropping.
3 Ruitenbeek (1989a)	$15b 3 Medicinal/genetic net	protection — would start to exhaust resources. The imputed benefit stream	does not report any foreign (explicitly non-use) WTP. However (2) set the	benefits for the creation of the Mantadia National	2 Total present value $1296bn over 3.6bn
4 Tobias and Mendelsohn (1991)	present value $7/ha over 126,000ha (park area) or 426, 000ha (with the additional	therefore represents the continued existence of resources).	existence value for the Brazilian Amazon at $30b, calculated using an arbitrary WTP figure	Park (Madagascar) Implicitly their valuation will reflect a total economic value	hectares = $360/ha relative to a net revenue from clear felling of $1000/ha. The implied

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
	buffer zone). This represents a		(observed from		
5 Pearce (1991 d)	minimum expected genetic value. Estimates depend on i) the probability of an area	An imputed value of the expected loss from flooding resulting from alternative land	various CVM studies), aggregated across the OECD adult population.	of the resource forgone. The survey revealed a	ratio of 0.36 will not be strictly representative since the calculation of
6 Schneider (1991)	yielding a drug base ii) the method of valuation iii)an assumed extent of rent capture	use from 2010 onwards: NPV of expected value of loss by 2040 is £1.6m	Donations to charitable funds may be one possibility	per household expected mean WTA of $108 per annum	total economic value is not necessarily based on the assumption of sustainable
7 Balick and Mendelsohn (1992)	by local authority. 4 Travel cost valuation of	($2.84m) or $23ha. Soil fertility maintenance.	to place CV evaluations in context; however dichotomy between	which is aggregated over the affected number of households	use. 4 Implied for Costa Rica
8 Pearce (1990a)	tourist trips to Costa Rica's Monteverde Cloud Forest. Average visitor valuation $35	Benefit imputed based on crop productivity decline from soil loss which would	observed reason for giving and actual use of funds. Problem of identifying	amounts to a necessary one time compensation of	12.5 which is the ratio of recreation value per hectare of protected area
9 Watson (1988)	(1988), producing a present value for trips assuming constant flows of $2.5m, or	take effect from 2010 onwards (the without project scenario) NPV £532,000	organizations involved uniquely in forest protection.	approximately $673,000 using a 10% discount rate and a 20	to the highest estimated price of land outside the park
10 Kramer et al (1993)	extrapolating for foreign visitors $12.5m. This gives a value per hectare in the reserve of $1250	($958,000) or $8ha. 5 6 Valuing carbon	3 Value of debt-for-nature swaps may provide	year horizon. 16 Categorized	7 On the basis of local medicinal plant harvesting

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
11 Guttierez and Pearce (1992)	relative to the market price of local non-reserve land of $30–$100/ha.	sequestration; crediting standing forest with damage avoided from adverse climate change: $1.2b–$3.9b/year,	an approximation of a WTP reflecting a non-use value. Varying implicit valuation of different sites is reflected in	annual non-market benefits of 51.5 million hectares of Mexican forest ($m):	only, the implied ratio of 1.04.9 Determination of
12 Pinedo-Vasquez et al (1992)	7 Sustainable harvesting of medicinal plants in Belize (*local market values alone*) net present value $3327 per ha compared	depending on assumptions of: i) Damage estimate per tonne carbon estimated	the price paid by conservation bodies involved. Some swap transactions have aimed to	Tourism 32.2 NTFP na Carbon 3788.3 Watershed	market prices in this study is uncertain (ie world or local) implied ratio 11.3.
13 Solorzano and Guerrero (1988)	to $3184 from plantation forestry with rotation felling.	range $5–13 tonne. ii) amount released, itself dependent on assumptions	preserve tropical forest ecosystems (see Chapter 6).	protection 2.3 Option value 331.7 Existence value 60.2	3 1.07 total project ratio or 1.94 from the perspective of Cameroon
	9 Forest production	of per hectare sequestration		Total 4214.8	when indirect project
14 Schneider (1992)	(Malaysia) $2455/ha compared to $217/ha from intensive agriculture.	and annual deforestation rates.	10 Foreign visitor's WTP for increased Lemur siting in the hypothetical Mantadia	Total should be regarded as a lower bound estimate.	adjustments are included.These include figures for project related aid flows
15 Grimes et al (1993)		8 14 Carbon storage	National Park (1991). Mean		and value for uncaptured
	3 Tourism value from the Korup $19/ha.	$1300–5700ha/year.	bid was $65 (conditional on seeing the same number of		genetic and watershed values.
16 Pearce et		11 Total carbon storage	Lemurs actually seen in		

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
al (1993)			the		
	10 Annual value of fuelwood to Malagasy households about $39 per annum.	value Brazilian Amazon $46bn.	Perinet Reserve where visitors were surveyed). At current visitation rates to		13 Implied ratio of 2
		13 Rio Macho Preserve,	Perinet, the new park would		
	15 Valuation of sustainable non timber harvest from 3 one hectare plots in the Ecuadorian Amazon, average net present value for the sites of $2306.	Costa Rica. Evaluates the replacement cost in terms of water services and energy generation resulting from reserve conversion to agricultural use.	generate a total additional WTP of $253,500 pa. This amounts to a present value of $2.16 million discounting over 20 years at 10%.		

Table 6.2 *Economic values — wetlands*

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
Ecosystem type: Wetlands	1 NPV per acre ($1990) from the preservation of the Hadejia- Jama'are floodplain, Nigeria.	1 Groundwater recharge function for surrounding areas, potentially measurable	Significant option values from future tourism, educational and scientific	7 Bintuni Bay mangrove ecosystem, Irian Jaya. NPV of	1 Benefit/cost ratio expressed in terms of the relative benefits accruing
Sources:		by either WTP or using costs	uses.	whole system ($1991	from alternative water use:

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
1 Barbier, Adams and Kimmage (1991) Agriculture 41Fishing 15 Fuelwood 7 Discounted at 8%	of groundwater depletion on local agriculture — ie a *production function* approach — as a minimum benefit	Existence values of wetland wildlife probably high although no explicit studies exist.	discount rate 7.5%) $961 – 1495m of which direct-use probably $152–534m.	$45 per 1000m^3 of water maintained in the floodplain as opposed to 4	cents per 1000m^3 from
	Other floodplain benefits:	approximation.		This value does not	diverted water.
2 Samples et al (1986b)	Livestock and grazing Non-timber forest products Tourism, Recreation (including	*Other important functions:* Flood control and storm	2 Some non-use values for wildlife (CVM estimates) $1990/annum/person:	account for the high *cultural value* placed on the bay by the Irarutu	4 From a similar analysis of the Ichkeul National
3 Costanza et al (1989)	hunting), Educational and scientific benefits (genetic and information value).	protection can in theory be approximated estimating alternative *preventative*	brown bear, wolf, wolferine (Norway) 15.0	tribe (10).	Park, Tunisia, direct-usebenefits amounted to $134 per 1000m^3
4 Thomas et al (1990)	3 Louisiana. WTP Present value at 8% ($1990) per acre.	*expenditure or replacement costs* for sea defences and dykes. In Malaysia the cost of	bald eagle (US) 12.4 emerald shiner 4.5 grizzly bear 18.5	compared to *negative returns* from diversionary use.	
5 Bergstrom et al (1990)	Commercial fishery 400 Fur trapping 190 Recreation 57	rock escarpments to replace eroded mangrove fringe is typically around $300,000/	bighom sheep 8.6 whooping crane 1.2 blue whale 9.3	Given the difficulty of generalizing with respect	
6 Thibodeau and Ostro (1981)	Storm protection 2400 Total 3047	km ($1990) (11). The same study quotes a 1987 EC estimate of the `inherent'	bottlenose dolphin 7.0 California sea otter 8.1	cost-benefit ratios are	to alternative uses for wetland areas informative
	5 Louisiana. WTP Present	value of mangrove	9 Revealed WTP (CVM)		difficult to provide.

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
7 Ruitenbeek (1991)	value at 8% ($1990) per acre Recreation 103	protection to Guyana as $4bn, though there is no indication of how the figure is	for preservation benefits of blanket bog area in Scotland (1990) (once-and-for-all	Where non-use values have been inferred from costs of imposing or	
8 Hamilton and Snedaker (eds) (1984) 00	6 Charles River, Massachusetts. Present value ($1990) per acre at 8%. Recreation 3400	derived. Nutrient cycling will normally have a measurable effect on	payment) Present value £164.68/ha (approx $296.50/ha) implicitly representing the discounted	agreeing land use constraints (the cost of which represents a discounted future benefit	
9 Hanley and Craig (1991)	Water supply 80,000	fishing and agricultural yields (in deltaic areas) the value of	future stream of user and non-user benefits. As such	stream), the implicit cost-benefit ratio will normally	
	8 Present value per acre (at	which might also be	the value is interpreted as		be at least I because the
10 Van Diepen and Fiselier (1990)	8%) of Mangrove systems. Direct use from fisheries, forestry and recreation. Trinidad $15,000	approximated by *replacement expenditures* on nutrients and compensating technologies.	an option value. (See Smith, 1987). 12 An average annual	compensatory payment from the recipient's perspective will have to be at least equal to the	
11 Fiselier (1990a)	Fiji $11,000 Puerto Rico $13,000	14 Sustainable charcoal production from mangrove	amount ($343/acre) paid (by the US Fish and Wildfowl Service in 1980)	perceived opporttunity cost.	
12 Danielson and Leitch (1986)	15 Tentative estimates of mangrove timber production in Sarawak of $123,217 per annum.	(Thailand) generates an annual national income of approx. $22.4m. Net profits are nearly $4000/ha for	to owners of Wetlands in Massachusetts for *preservation easements,* can be taken to represent a		

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
13 Turner and Brooke (1988)		forests with average productivity of 230m^3/ha. 15 Marine fisheries	minimum option value for the ecosystem in an unaltered state. Similar conclusions could be		
14 McNeely and Dobias (1991)	maintenance worth $21.1 million per annum.	inferrred by looking at the average value of *management agreements* negotiated between			
15 Bennet and Reynolds (1993)	conservation bodies and land owners in the UK. Such an *alternative cost approach* has revealed a value of £70/ ha/per annum for coastal marshland.				

Table 6.3 *Economic values — rangelands*

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
Ecosystem type: Rangelands (semi-arid) and Wilderness	1 Wildlife tourism. Viewing value of elephants in Kenya $25m/per annum.	*Indirect benefits from sustainable wildlife management:*	3 Beneficial use project for Khao Yai National Park surveyed user WTP for continued existence of		2 Ratio of wildlife tourism revenue per hectare ($40) to income from extensive pastoralism
areas	The same study gives an	*Distribution of benefits to local*	elephants at approx $7.		($0.80) 50. This ratio has
	indication of the extent of	*communities* as a result of	Under certain assumptions		probably increased

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
Sources: 1 Brown and Henry (1989)	revenue forgone through sub- optimal park entrance pricing. A rough WTP survey revealed a potential consumer surplus as	sustainable wildlife management schemes. 7 The Nyaminyami Wildlife Management Trust,	of population and park use, the option and existence value of Khao Yai to Thai residents (for elephant		significantly due to increasing value added in tourism.
2 Western and Thresher (1973)	high as $25m/per annum (a sum almost 10 times the value of poached ivory exports and at least a 10% increase in actual	Zimbabwe channelled approx Z$198,000 (1989) of wildlife revenues into local projects for health, housing,	preservation) may be as high as $4.7m/year. The extent of existence		4 Ratio of value of wildlife production (Z$4.20/ha) to Cattle Ranching (Z$3.58/ha) in
3 Dobias (1988)	expenditures). Since people were only asked their WTP to preserve elephants, consumer	education and recreation. In addition the project was able to compensate local	values might be approximated from the value of *vicarious tourism* —		Zimbabwe 1.17. Calculation based on *economic rates of return* (as
4 Child (1984) (1990)	surplus for all wildlife viewing is presumably higher.	farmers for any damage incurred and offer cropped wildlife products for sale	the consumption of books, films and TV programmes — particularly in developed		opposed to financial rates), and accounting for the relative environmental
5 Coulson (1991)	4 *Wildlife utilization:* Non-consumptive game viewing, lightly consumptive safari	locally at subsidized prices. Direct and indirect provision	countries, or from observed *charitable donations* to organizations involved in		costs would in certain areas of the country produce ratios of between

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
6 Dept of National Parks, Zimbabwe (1991)	hunting and live animal trade, consumptive meat and hide production. *Zimbabwe illustrative examples: Non-consumptive use:* Direct and	of employment. Improvements in local infrastructure and potential increases in land and	wildlife preservation. More crudely we might extrapolate on the basis of WTP information of visitors to wildlife sites in substitute		2 and 5. 8 Provides Present values for returns from game viewing combined
7 Jansen (1990)	indirect income accruing to the Matusadona National Park (1991) US$10.3m, 66% of	property values. Significant saving in the	countries like Kenya. In 1990 56% of overnight		with some form of elephant cropping and for viewing alone in Botswana
8 Barnes (1990)	which foreign currency (5). *Safari hunting:* Value for foreign visitors in 1990 US$9m of	hidden costs of land degradation and soil erosion arising from agricultural	visitors to wildlife areas in Zimbabwe were foreign, of which 26% originated in		(1989). The ratio of the former to the latter range from 2.63 to 1.8
9 Imber (1991)	which, value of trophies US$4m (6). *Consumptive value:* Zimbabwe	production in marginal areas. The role of elephants as	Europe or North America (approx 151,000 visitors). Assuming 50% of these		(depending on whether a 5 or 15 year horizon is considered)
10 Norton- Griffiths and Southey (1993)	estimates it makes $4.7m/ annum from the sale of elephant goods and services, a return of $75/km^2 over approx	keystone species diversifying savannah and forest ecosystems.	visitors reveal a similar WTP *in addition* to entry fees (in much the same way as in (1)ie a $100 permit for		demonstrating the earning potential of consumptive uses. Comparison with the economic rate of return

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
11 Moran (1994)	74,000km^2 of elephant habitat. The proportion attributed to sale of goods has fallen significantly since the imposition of an international ban on ivory sales.	Value added retained in the host country consists of net revenues accruing to: local airlines, tour operators, hotels, transport and cottage industries.	elephant preservation), extra revenue generated might amount to $7.5m per annum. 9 CV study preserve the Kakadu Conservation Zone		from cattle production on a per hectare basis could show ratios similar to those in Zimbabwe. 10,11 Implicit tourist use to opportunity cost ratio
	10 Reports an opportunity cost estimate for Kenyan protected areas. Forgone revenues average $34 per conserved hectare per annum. However (11) elaborating on (1) reports an expected mean WTP per day by tourists of $72 for park maintenance. Depending on the visitation scenario (number of days on safari) this translates to a per hectare consumer surplus range of $16–$157.		(from mining development) revealed that Australians were willing to pay A$124/ annum for ten years to avoid a major impact scenario and A$53 to avoid the minor scenario. Extrapolated to the whole population produced a total WTP range of A$650m- $1520m or a present value at 5% of between A$1m/ha and A$2.3m/ha over 5000ha.		range 0.5–4.6

Table 6.4 *Economic values — marine systems*

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
Ecosystem type: Marine/Coastal systems, Heritage sites	1 Estimating the socio-economic effect of the Crown of Thoms starfish on the Great Barrier Reef. A travel cost	2 Estimates provided for the Galapagos National Park, Ecuador: $/ha/year Maintenance of Biodiversity	2 Option value for the Galapagos National Park set arbitrarily at $120/ha/ year which is the	2 Total annual monetary returns from direct and indirect use approx	3 Cahuita National Park ratio 9.54*. 4 Ratio of total (direct
	approach provided estimates of	4.9	approximate sum of direct	$120/ha. In present	and indirect) benefits to
Sources: 1 Carter et al (1987)	consumer surplus of A$117.5m/ year for Australian visitors and A$26.7m/year for international visitors. The study showed that	Value of fish breeding (nursery function) 0.07 (applicable to 430,000ha of marine zone).	and indirect use values from the park. The uniqueness of the Galapagos ecosystem	value terms this represents $2400/ha (at 5% discount rate) or almost $2.8bn for	total cost 11.5*. *A conventionally assessed
Hundloe (1990)	tourism to the reef is valued (in NPV terms) over and above current expenditure levels by	Watershed and erosion prevention functions 0.3 (applicable to terrestrial area	suggests that existence values are likely to be significant.	the entire study area. 8 Reports a mean	ratio rather than one based on opportunity cost.
2 de Groot (1992)	more than $Albn.	of 720,000ha).	5 Describes a CV survey	annual WTP estimate of $27 for diving in the	
	2 Total direct use valued at $53/		to value visibility	Bonaire Marine Park	
3 Marcondes (1981)	ha/year, comprising ($/ha/year): recreational use 45 food/nutrition 0.7		improvements at the Grand Canyon (from reduced sulphur dioxide	At current visitation rates estimated consumer surplus is	

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
4 Posner et al (1981)	Raw materials for construction 5.2 Energy resources 1.5 Ornamental resources 0.4		emissions). Mean bid ($1990/person/year) $27. A high level of familiarity	$325,000.	
5 Schulze et al (1983)	Biochemical and genetic resource values are also thought to be significant though no estimates are		may explain the high value respondents seem to have been willing to pay in this		
6 Hausman, Leonard, McFadden (1992)	provided. Provision of employment directly or indirectly related to the National Park is a considerable benefit to the Galapagos economy (60% of		study (compared to bids for endangered species — see Table 5.3). Higher WTP bids in habitat valuation studies have		
7 Carson et al (1992)	2500 workforce). Tourism is the most important activity, contributing an estimated $26.8m to the local economy.		generally revealed a preference for protection of a perceived array of benefits rather than for a targeted species. As with other CV studies the Grand Canyon case has been the subject of much debate, particularly with respect to the levels of information and framing (hypothetical) bias (see Schulze et al, 1981).		
8 Dixon et al (1993)	3 A form of travel cost appraisal of the recreational value of the Cahuita National Park,		7 Estimate median per household WTP of $31 as a one-off amount to		

Value category:	Direct use	Indirect use	Non-use values option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use)/opportunity cost ratio
	Costa Rica. Consumer surplus estimates were derived from observed wage equivalent travel time net of transport costs multiplied over a visitor population. The resulting benefit-cost ratio demonstrated that the park is economically beneficial.				
	4 Conventional benefit-cost analysis of the Virgin Islands National Park, St Johns, identified significant direct and indirect benefits associated with the park, particularly tourist expenditure and the positive effect on land values in proximity to the designated area. Little information is available on the environmental effects of alternative land uses or the extent of visitors' consumer surplus. Total benefit ($1980) approx $8295/ha over approx 2820ha of National Park on St Johns.		prevent future oil spills. Aggregating over affected households derives an estimate of $2.8 billion as the total lost passive-use values as a result of the *Exxon Valdez* oil spill.		
	6 Recreation demand study to value recreation use loss caused by the *Valdez* oil spill in Alaska; about $3.8m				

ISSUES ARISING FROM THE EMPIRICAL ESTIMATES

Tables 6.1 to 6.4 suggest a number of issues for discussion.

What is being valued?

So far we have used the term `biodiversity' for the object of valuation. But it will be evident from Tables 6.1–6.4 that what these economic studies are measuring is the economic value of `biological resources' rather than their diversity. The reality is that most of the studies relate to the value of biological resources. How far they are capturing diversity is difficult to assess. Those studies looking at the valuation of habitats (eg the Kakadu study) may well be capturing perceptions of diversity — ie valuations may be high simply because the area is known to be rich in diversity. Diversity valuation requires some idea of the WTP for the *range* of species and habitats rather than the specific biological resources they happen to support. Valuing diversity as such will be far more complex. Contingent valuation approaches offer the most promise since individuals can be presented with different ranges of species/habitats to see which they prefer. But information will be paramount: many life scientists believe that this diversity is fundamental to human well-being. Others argue that we simply do not know what the functions of diversity are. As such, individuals may not be informed of the potential value of diversity. Travel cost and discrete choice studies might also be used for diversity valuation if it is possible to look at choices between alternatives that vary in their degree of diversity.

Transferability

The extent to which the various estimates can be transferred from one context to another is open to question. Many of the valuations are specific to the areas studied. For example, it is unlikely that values based on contingent valuation will be transferable. Those obtained by an assessment of the local market — eg for medicinal plants — will reflect supply and demand in that particular context. Those conditions will vary in other contexts.

A second problem of transferability arises in the context of generalizing values to wider areas. For example, the study by Peters, Gentry and Mendelsohn (1989) on returns to the sustainable use of land in Peru cannot be used to argue that such values are typical of *all* tropically forested sites, even in the same region. This is because (a) the values of the non-timber products in question will be related to the existence of a market place. The further from the market, the lower such values can be expected to be, and (b) the expansion of non-timber product supply across all feasible sites would result in price falls for the commodities in question. As it happens, the Peters et al estimates have been severely criticized. They are for the value of the *inventory* of non-timber products rather than the *sustainable flow* of products actually used. The latter might be only three per cent of the inventory value (Padoch and de Jong, 1989). This disparity highlights some of the serious methodological differences underlying the received view of non-timber product exploitation. Godoy, Lubowski and Markandya (1993), point out that such studies lack consistency in the examination of extraction costs, methods for ascertaining household uses, and, in determining the correct `forest gate' or alternative shadow price to be assigned to marketable quantities. Indeed the whole exercise of valuation seems in most cases to have been an afterthought. Even more serious, is the observation of a singular lack of evidence that valuation estimates are compatible with sustainable extraction. Hall and Bawa (1993) discuss methods for assessing biological sustainability of plant extraction, but few studies attempt to gauge the sustainable hunting yields. In the absence of more detailed analysis therefore, sustainability of forest use is at best case specific. When extraction is known to be non-sustainable, this can be indicated in an appraisal by deduction of depletion premium from the value of forest products. This would depend on how long the present extraction can continue and on the discount rate (Godoy, Lubowski and Markandya, op cit).

Although fraught with difficulties, the issue of transferring benefit estimates is appealing. Avoidance of a full-blown benefit study may represent a considerable resource saving for funders and agencies implementing environmentally sensitive projects. In developed countries, such savings are motivating considerable interest in an analysis of appropriate conditions for transferring estimates, and the establishment of a `values library' of off-the-shelf estimates (Boyle and Bergstrom, 1992).

BENEFITS TRANSFER

What are the necessary conditions for using the economic value of an environmental change at one site as a surrogate for a similar change elsewhere? Is it always necessary to initiate a new study in a project area to determine how the well-being of individuals is potentially affected by an environmental change?

Suppose that a development project would result in the destruction of a certain number of hectares of wetlands, and an analyst wished to estimate the economic value of the environmental losses associated with this proposed project. Rather than attempt to undertake a new study at the site of the proposed development, the analyst could identify previous studies that had estimated the economic value of wetlands, and then assume that the loss of a hectare of wetland at the proposed development site would be the same as (or similar to) this previous estimate. Such an approach has been termed `benefit transfer' because the estimates of economic benefits are `transferred' from a site where a study has already been done to the site of policy interest. The benefits transferred from the study site could have been measured using either the direct or indirect valuation techniques outlined above. More formally, suppose the willingness to pay of household i for a change from an initial environmental endowment Q_0 to an improved environmental endowment Q_1 is given by:

$$WTPi = f(Q_1 — Q_0, P_{own,i}, P_{sub,i}, S, E_i) \quad (1)$$

where

P_{own} price of using the environmental resource (`own price')

P_{sub} price of substitutes for use of the environmental resource

S_i socioeconomic characteristics of household i

Benefit transfer requires three steps.

1 We must find (a) a study where this demand relationship has been estimated for an existing site; and (b) where values for Q_1, P_{own}, P_{sub}, and S exist at the new site that we wish to value;

2 We must determine the geographic area over which households will benefit from the change in environmental quality;

3 We must substitute the values of the independent variables for the households at the new site that is being valued into (1) to calculate the benefits to household i at the new site. Then the analyst must aggregate these estimates for all households affected in order to obtain the aggregate benefits at the site.

It is not necessary that an analyst be restricted to the use of just one site as the source of information to be transferred to the new site. Information could be obtained from several sites and summarized for transfer to the new site. For instance, in the example above the analyst could take the average estimate of the value of a hectare of wetland from existing studies. A more sophisticated approach would attempt to explain the determinants of variation in parameter estimates from existing study sites, and then use this model and values of the independent variables (the determinants of the variation) from the new site to estimate the benefits (value) at the new site. Such *meta analysis* techniques, which test consistency of valuation estimates using a data base of similar studies are a useful precursor to benefits transfer. The process does, however, require the availability of a bank of reliable estimates.

Most of the existing applications of benefit transfer methods in non-market valuation have attempted to estimate the recreational benefits of new projects or opportunities. The advantages of the approach are basically that estimates of economic benefits can be obtained more quickly and cheaply than if a new valuation study is attempted requiring primary data collection and often taking much time to complete.

There are three approaches to benefits transfer:

1 *Transferring mean unit values* — here we assume that the change in well-being experienced by the average individual at the existing sites is equivalent to that which will be experienced at the new site being valued. The previous studies are used to estimate the consumer

surplus or average WTP of individuals engaged in, say, recreational activities of various kinds. These WTP values of a day spent by a person in a specific type of recreational activity at the existing sites are multiplied by the number of days of such activity forecast to change or occur at the new site as a result of the environmental change, to obtain an estimate of the aggregate economic benefits from the recreational activity at the new site.

Table 6.5 shows a summary of unit values of WTP for days spent in various recreational activities obtained from 287 existing studies. These unit values are the estimated amounts that individuals would be willing to pay over and above their current expenditures in order to ensure the continued availability of the resource for recreational use. Both travel cost models and the contingent valuation method were used to obtain these estimates.

The problem with this approach is that individuals at the new site, for a variety of reasons, may not value the recreational activities at the new site the same as the average individual at the existing sites studies on which the unit values are based. More sophisticated benefit transfers can instead be attempted as below.

2 *Transferring adjusted unit values* — here the mean unit values of the existing studies are adjusted before transferring to the new site. The unit values can either be adjusted for any biases that are thought to exist, or they can be adjusted in order to reflect better the conditions at the new site. Potential differences that should be looked for between the existing and new site are:

Table 6.5 *Net economic values per recreation day reported by TCM and CVM demand studies from 1968 to 1988, United States (third quarter 1987 dollars)*

Activity	Number of estimates	Mean	Median	Standard error of mean	95% confidence interval	Range ($)
Total	287	$33.95	$27.02	$1.67	$30.68 – 37.22	3.91 – 219.65
Camping	18	19.50	18.92	2.03	15.52 – 23.48	8.26 – 34.89
Picnicking	7	17.33	12.82	5.08	7.37 – 27.29	7.05 – 46.69
Swimming	11	22.97	18.60	3.79	15.54 – 30.40	7.05 – 42.94
Sightseeing and off-road driving	6	20.29	19.72	3.73	12.98 – 27.60	10.33 – 31.84
Boating, motorized	5	31.56	25.67	10.36	11.25 – 51.87	8.27 – 68.65
Boating, nonmotorized	11	48.68	25.36	15.85	17.61 – 79.75	10.26 – 183.36
Hiking	6	29.08	23.62	5.82	17.67 – 10.49	15.71 – 55.81
Wintersports	12	28.50	24.39	4.48	19.72 –	11.27 –

Activity	Number of estimates	Mean	Median	Standard error of mean	95% confidence interval	Range ($)
					37.28	66.69
Resorts, cabins, and organized camps[a]	2	12.48				3.91 – 19.93
Big game hunting	56	45.47	37.87	3.47	38.67 – 52.27	19.81 – 142.40
Small game hunting	10	30.82	27.48	3.51	23.94 – 37.70	18.72 – 52.04
Migratory waterfowl hunting	17	35.64	25.27	5.87	24.13 – 47.15	16.58 – 102.88
Cold water fishing	39	30.62	28.49	3.24	24.27 – 36.97	10.07 – 118.12
Anadromous fishing[b]	9	54.01	46.24	11.01	32.43 – 75.59	16.85 – 127.26
Warm water fishing	23	23.55	22.50	2.46	18.73 – 28.87	8.13 – 59.42
Salt water fishing	17	72.49	53.35	14.05	44.95 – 100.03	18.69 – 219.65
Nonconsumptive fish and wildlife	14	22.20	20.49	2.30	17.69 – 26.71	5.27 – 38.06
Wilderness	15	24.58	19.26	6.10	12.62 – 36.54	8.72 – 106.26
Other recreation activities	9	18.82	16.06	3.65	11.67 – 25.97	6.81 – 43.39

Source: Walsh, R G, Johnson, D M and McKean J R (1992) `Benefit Transfer of Outdoor Recreation Demand Studies, 1968—1988, *Water Resources Reserch,* vol 28, no 3, March (Special Section: Problems and Issues in the Validity of Benefit Transfer Methodologies).

— differences in socioeconomic characteristics of households;

— differences in the environmental change being looked at; and

— differences in the availability of substitute goods and services.

3 *Transferring the demand function* — instead of transferring adjusted or unadjusted unit values, the entire demand function estimated at existing sites could be transferred to the new site. More information is passed over in this way. As an example, for a zonal travel cost model, the demand function might be of the form (Loomis, 1992):

$$X_{ij}/POP_i = b_0 — b_1 C_{ij} + b_2 Time_{ij} + b_3 Psub_{ik} + b_4 I_i + b_5 Q_j \quad (2)$$

where

X_{ij} number of trips from origin i to site j

[a] Resorts were 1.83% valued at $19.93 per day; seasonal and year-around cabins were 3.06% valued at $3.91 per day; and organized camps were 1.79% valued the same as camping.

[b] Anadromous fishing estimates included in cold water fishing. Estimated as roughly 5%

POP_i	population of origin I
C_{ij}	travel costs from origin i to site j
$Psub_{ik}$	a measure of the cost and quality of substitute site k to people in origin I
I_i	average income in origin I
Q_j	quality of site j for recreational uses.

The approach requires that estimates of the parameters b_0, b_1, b_2, b_3, b_4, and b_5 are found from existing studies. Data is then collected on

1. population of zones `around' the new site;
2. travel costs from these zones to the new site;
3. the cost and quality of the alternative recreational sites available to people living in the zones designated to be around the new site;
4. the average income of people in these zones; and
5. a measure of the quality of the new site for recreational uses.

The values of these independent variables from the policy site and the estimates of b_0, b_1, b_2, b_3, b_4, and b_5 from the study site would be replaced in the travel cost model (2), and this new equation could then be used to estimate both the number of trips from the designated zones to the new site and the average household willingness to pay for a visit to the new site (see Cicchetti, Fisher and Smith (1976) for an early example).

The use of benefit estimates is still in its infancy, and so procedures for dealing with problems common to most studies have not been standardized. Moreover, the paucity of valuation estimates in developing countries is such that value transfers could at best be international rather than on a (preferable) regional or local level. Additional socio-economic and cultural barriers to direct transfer of estimates are a further important qualification for any resulting appraisal. For the time being therefore, the practice outside countries with a sufficient bank of benefits research is likely to be restricted.

THE ECONOMIC VALUE OF MEDICINAL PLANTS

The potential pay-off from a `blockbuster' drug provides a compelling argument for the identification and preservation of the world's most species rich ecosystems. Yet quantitative assessment of the likelihood of such returns is at best speculative, with some debate surrounding the merits of natural product screening relative to biotechnology and chemical synthesis. Plant species are used for medicines in two ways: (a) major commercial use, whether by prescription or over-the-counter sales, and (b) as traditional medicines which may or may not attract a market price. In the rich world, perhaps 25 per cent of all medical drugs are based on plants and plant derivatives. In the poor world the proportion is closer to 75 per cent (Principe, 1991). Clearly, both uses have an economic value, but what little work there is on the valuation of medicinal plants tends to focus on category (a). This should not obscure the likelihood that local willingness to pay for traditional medicines may be substantial. However, the difficulties encountered in accurately measuring the returns to local non-timber product uses apply equally to the investigation of ethnopharmacological uses. We therefore focus on the economic value of commercial drugs based on plants.

Substitution

While some medicinal plant resources have not been reproduced synthetically (*digitoxin,* for example), and others have been reproduced but are less efficient than the original material (synthetic vincristine from *Catharanthus* for example), many synthetic substitutes do exist. Two considerations are relevant. First, can biological diversity conservation be justified on the basis of the need for `one off' exploitation before the material is synthesized, together with continued exploitation of plants which cannot be

synthesized? Second, are future drugs more, or less likely to be manufactured from plant-based materials? The answer to the former question depends on the economic values that can actually be ascribed to the biodiversity resource, and on the answer to the second question. The answer to that appears to be uncertain. Principe (1989) reports on a UN International Trade Centre study which suggests that pharmaceutical companies have shown a decreasing interest in the development of new botanical products in favour of molecular biology and biotechnology applications to micro-organisms. Processing plant genetic material is time-consuming and expensive, and simple comparative rates of return are higher from other routes. On the other hand, others in the industry appear to believe that plant-based resources will re-emerge, facilitated by advances in screening processes, options for tapping into existing unexplored indigenous uses, and the potential to franchise out much of the screening process to host countries. This increased *in situ* capability appears to have been an important factor in the much-publicized prospecting agreement between Merck and the Costa Rican Instituto Nacional de Biodiversidad (Laird, 1993). Such agreements epitomize the potential for mutual gain and have focused considerable attention on the necessary national and international institutional requirements for maximizing returns to all parties.

Principe (1989) reports several reasons why research based on micro-organisms has limitations. The most important are (a) the steps of identifying the chemical structure required to achieve a given effect and creating a proper genetic code structure are the most difficult stages of drug development, and these are not helped by micro-organisms rather than plant-based genetic material, and (b) genetically engineered micro-organisms can, so far, substitute for only some of the plant-based chemicals. Indeed, Principe reports that the vast majority of plant-based chemicals have not been successfully synthesized.

The future of drug development may also be more, rather than less, dependent upon plant genetic material in light of the fact that plant-based research has gone in cycles. Findeisen (1991) reports that many thought that plant-based drug resources were exhausted in the early part of this century. The role of plants was, however, revived in the 1940s and 1950s with the discovery of the Vinca alkaloids (*Catharanthus rosea*) and reserpine (*Rauwolfia serpentina*). When the screening programmes at the National Cancer Institute and elsewhere in industry failed to come up with significant discoveries, the industry lost interest and screening programmes were effectively halted in the 1970s. The disinterest was compounded by the difficulties of plant-based drugs. The patent has to relate to the process of manufacture or to some unanticipated use value. Natural compounds *per se* cannot be patented. Thus, the Mexican government took control of Diosgenin resources in order to capture the rent from the production of *Dioscorea,* the main source of steroids in the early days of that drug. Attempts at the monopoly pricing of the resource forced pharmaceutical companies to search for synthetic substitutes. The case illustrates the problems of patenting and the problems facing countries that do seek to capture rents from biodiversity.

Some revival of interest in plant-based approaches in the last five years is accounted for by new techniques of purifying, analysing and assaying plant samples, including the use of robots for continuous assay of material. It is reported that among others, the National Cancer Institute, Monsanto, Smith Kline, Merck and Glaxo have revived plant screening programmes (Reid et al, 1993). Affymax and Shaman are new US companies developing drugs solely from natural products, and with a lot of emphasis on traditional medicines. The other main source of a revival in interest in medicinal plants is consumer demand for `natural products'. Whilst consumers are unlikely to express a concern about the source material for major life saving drugs, they do express a significant concern about the sources of over-the-counter drugs and cosmetics, as the success of some natural products shops reveals.

Clearly then, medicinal plant values are relevant to use value arguments for conserving biological resources in the developing world. How far they have relevance in justifying conservation of biodiversity as such is more problematic. Some commercial sources doubt that genetic engineering of micro-organisms will totally displace plant-based research. This would suggest an insurance argument for conserving at least minimum diversity based on arguments related to the option and quasi-option values of the resource. These arguments are all the more powerful because of the extremely limited knowledge that exists about the medicinal properties of plants. But even here there are qualifications. There is not an equal likelihood that each species will generate a chemical suitable for manufacture into a beneficial drug. Some plants are simply more likely to contain the right genetic make-up than others. How far diversity needs to be protected to capture these option values is open to question. It may be feasible to focus on a

`genetic slice' of biodiversity and pay little attention to the broad spectrum of species. Offsetting this possibility is the argument that the genetic slice will itself depend for its existence and functioning on the broad diversity of species.

Evenson (1991) addresses this question to an extent. He distinguishes between two fundamental values of genetic resources as producer goods: one in the general strategic search for new resources which justifies the maintenance of most materials, and another in the specialized search for genetic material to meet specific needs, which justifies the collection and preservation of `fringe' genetic resources. His calculations for rice suggest that if there is an economic case for maintaining an *ex situ* collection, the case for maintaining a near complete collection is stronger.

Overall, then, ascribing an economic value to medicinal plants can be done on two bases. The first relates to existing use values which, in turn, are for commercial drugs and for traditional medicine. The second relates to the *option value* of the plants, ie the extent to which conservation is required to protect *future use values.* Option values, in turn, are critically dependent upon the future of research in the medicinal drugs sector with respect to the base materials that are likely to be used.

THE ECONOMIC VALUE OF PLANT-BASED DRUGS

Ideally, what is required for economic valuation purposes is some idea of the ruling prices for plant genetic material and elasticities of demand by drug companies for that material. Given the availability of synthetic substitution as an alternative technology for some drugs, it seems clear that the demand elasticity will be high for those drugs, but fairly low for plant-based material that cannot, so far anyway, be synthesized. Drug companies today tend to use specialist plant gathering agencies (botanical gardens in the USA and a private company, Biotics, in the UK). In turn the gathering agencies use local institutions and people to engage in actual collection and shipping. Payment to the gathering companies is by contract or weight of material, but there are examples of agreements involving royalties in the event of successful exploitation. Thus, Biotics has royalty agreements with the companies it supplies and, in turn, those royalties are divided between the company and the source countries. To this end, these agreements already provide for the sharing of rents in the way clearly intended by the Rio Biodiversity Convention. Findeisen (1991) reports that royalties are usually negotiated on the basis of the value of the drug to the drug company, with royalty figures being in the range 5–20 per cent. But royalties are more readily negotiated for plant material to be used in a drug that is near to being marketed. Material that is destined for screening for longer term development is likely to attract low royalty agreements or simple one-off fees. Other companies have straight retainer agreements with botanical gardens and no royalty agreements. In the model used later, we therefore assume that a royalty rate of 5 per cent is applied to any plant material that results in the development of a successful drug.

Economic valuation to date has been fairly speculative but illustrative of the orders of magnitude involved. There are several ways in which to approach valuation:

- by looking at the actual market value of the plants when traded;
- by looking at the market value of the drugs of which they are the source material;
- by looking at the value of the drugs in terms of their life-saving properties, and using a value of a `statistical life'.

If we do not take into account the prevailing institutional capability to capture the values in discoveries as implied in the last two points, the result will be exaggerated valuations for the host country. As Ruitenbeek (1989) notes, the economics of invention reveals that income realized by inventors is considerably less than the ultimate value to society of the product, because the traits associated with the ultimate products have a very low degree of appropriability. This is true with respect to the countries providing niches to the diverse flora and fauna where the discoveries have to be made. This aberration in rent appropriation becomes even more blurred when the assumptions of ignorance, uncertainty, essentiality, and substitutability about medicinal plants enter the analysis. This implies that a factor representing the institutional framework should be applied to the ex-post discovery valuation. This factor will depend on the existence of the licensing structure in the host countries; whether research conducted in the host country

causes other leakages in the economy; and whether the ability exists domestically to carry out the research. Thus this factor is expected to be low in tropical low income economies. In Ruitenbeek's terms:

$$CPV = a.EPV$$

where CPV is capturable production value, EPV is expected production value, ie the patent value of one discovery. The fact that *a* tends to be low explains why developing nations feel that the benefit of their efforts to conserve biodiversity is captured more by others. That is, *a* can be thought of as a *coefficient of rent capture.* One purpose of the Rio Biodiversity Convention is to raise the value of *a.*

MODELLING THE ECONOMIC VALUE OF MEDICINAL PLANTS

We are now in a position to develop a simple model for determining the medicinal plant value of a unit of land as biodiversity support. The approach is fraught with difficulties given the considerable data deficiencies, but it is worth pursuing.

For any given area, say a hectare, there will be some probability, *p,* that the biodiversity `supported' by that land will yield a successful drug *D.* Let the value of this drug be $V_i$ *(D)*, where subscript *i* indicates one of two ways of estimating the value: the market price of the drug on the world market (*i* = 1), or the `shadow' value of the drug which is determined by the number of lives that the drug saves and the value of a statistical life (*i* = 2). Since there are many other factors of production producing value in the drug, let *r* be the royalty that could be commanded if the host country could capture all the royalty value. Finally, let *a* be the coefficient of rent capture discussed previously. Then, the medicinal plant value of a hectare of `biodiversity land' is:

$$V_{mp}\,(L) = p.r.a.Vi\,(D)$$

We consider each element of this equation in turn.

Probability of success — Principe (1991) estimates that the probability of any given plant species giving rise to a successful drug is between 1 in 10,000 and 1 in 1000. These estimates are based on discussions with drug company experts. Estimates of the number of plant species likely to be extinct in the next 50 years or so vary, but a figure of 60,000 is widely quoted (Raven, 1988). This suggests that somewhere between 6 and 60 of these species could have significant drug values. Put another way, if biodiversity use was favoured over alternative land uses, the realized benefit as far as medicinal drugs are concerned would be the economic value of these 6–60 species.

The royalty — Based on the observation that existing royalty agreements involve royalties of 5–20 per cent, but with a low figure for drug development some way into the future, we adopt a value of r= 0.05.

Rent capture — If host countries could capture rents perfectly then *a* =1. Ruitenbeek (1989a) suggests that rent capture is likely to be as low as 10 per cent in low income countries. Hence a range for *a* is *a* =0.1 to 1.0.

The value of drugs — Table 6.6 summarizes some estimates of the value of successful drugs. The method of valuation is important because it affects the size of the estimate significantly. The valuation based on life-saving properties gives the highest values, using the value of a `statistical life' of $4 million (Pearce, Bann and Georgiou, 1992). Market values of plant-based drugs give lower values, and the actual traded price of the plant material the lowest value of all. The price of drugs reflects, of course, many more things than the cost of the plant source material. In that respect, the drug price grossly overstates the value of the plant. Equally, market prices understate true willingness to pay for drugs: there will be individuals who are willing to pay more than the market price for a given drug. Indeed, since the evidence suggests that such drugs tend to be price inelastic, this `consumer surplus' element could be substantial. While there is no empirical basis for supposing that the consumer surplus element exactly offsets the overstatement in the price estimate, the two factors do work in opposite directions.

In the 1980s only about 40 plant species accounted for the plant-based prescribed drug sales in the USA. Thus, on the basis of prescription values only (see Table 6.6), each species was responsible for $11.7 billion/40 = $290 million on average. Since all life-saving drugs would be on prescription, use of the value of avoided

Table 6.6 *Some values of plant-based pharmaceuticals*

	$ billion 1990 prices (bracketed numbers refer to the year to which the estimate relates)		
	USA	OECD	WORLD
Market value of trade in medicinal plants	5.7 (1980)	17.2 (1981)	24.4? (1980)
Market or fixed value of plant-based drugs on prescription	11.7 (1985)	35.1 (1985)	49.8? (1985)
	15.5 (1990)		
Market value of prescription and over-the-counter plant-based drugs	19.8 (1985)	59.4 (1985)	84.3 (1985)
Value of plant-based drugs based on avoided deaths: anti-cancer only	120.0	360.0	
+ non cancers	240.0 (1985)	720.0 (1985)	

Notes: Ratio of OECD to USA taken to be 3.

`Value of a statistical life' taken to be $4 million in 1990 prices.

Lives saved taken to be 22,500–37,500 per annum in USA. Average is taken here, ie 30,000. Multiply OECD by 1.4 to get world estimates.

Source: Adapted with modifications from Principle (1989). See also Principle (1991).

deaths suggests a value per plant of $240 billion/40 = $6 billion per annum. Clearly some species were far more valuable than others, but taking the average it is possible to get some idea of the lost pharmaceutical value from disappearing species. If there are 60,000 species likely to be unavailable for medical research, and the probability that any given plant will produce a marketable prescription drug is 10^{-3} to 10^{-4} then, taking a mean of 5.10^{-4} and applying it to the 60,000 estimated losses means that 30 plant-based drugs will be lost from species reduction. On market-based figures, the *annual* loss to the USA alone would therefore be 30 × $292 million = $8.8 billion, and to OECD countries generally perhaps $25 billion. In an update, Principe (1991) suggests that USA 1990 prescription plant-based medicines had a retail value of $15.5 billion, which would raise the value per plant to $390 million. As a benchmark, the GNP produced in the whole of Brazilian Amazonia is some $18 billion per annum. On the `value of life approach' the *annual* losses would be 30 × $6 billion = $180 billion for the USA, and over $500 billion for the OECD countries generally. However, these figures assume that substitutes would not be forthcoming in the event that the plant species did become extinct.

The value of land for medicinal plants

Using the previous estimates it is possible to arrive at an estimate of the value of a `representative' hectare of land. The model can now be written:

$$V_{mp}\,(L) = \{N_R\, p.r.a.\, V_i/n\}/H \text{ per annum}$$

where the new notation is:

N_R = number of plant species at risk

n = number of drugs based on plant species

H = number of hectares of land likely to support medicinal plants and

N_R = 60,000

p = 1/10,000 to 1/1000

r = 0.05

a = 0.1 to 1

V/n = 0.39 to 7.00 billion US$

H = 1 billion hectares, the approximate area of tropical forest left in the world.

The resulting range of values is from $0.01 to $21 per hectare. If a = 1 at all times, then the range is $0.1 to $21 ha. Clearly, the lower end of the range is negligible, but the upper end of the range would, for a discount rate of 5 per cent and a long time horizon amount to a present value of some $420 ha.

Other estimates of medicinal plant values

Ruitenbeek (1989) suggests an annual value of $85,000 (£50,000) for a = 1 for the Korup rainforest. The relevant area is either 126,000 per hectare (the central protected area) or 426,000 ha (the central area plus the surrounding management area), so that per hectare values would be $0.2 to $0.7 per hectare per annum, very much in keeping with the lower end of the range obtained from our own model.

In a study of medicinal plant harvesting in Belize, Balick and Mendelsohn (1992) estimate the local willingness to pay for land. Their annual net revenues are $19–61 per hectare. These values are not directly comparable to the estimates obtained above since they relate to local medicinal plant use rather than the `global' commercial values to the OECD countries. It is significant, however, that they just overlap the upper range of the global values obtained above ($21 per hectare). Note also that such local values would be quickly depressed if very large tracts of land were devoted to medicinal plants, whereas the global values obtained here would be fairly invariant with supply since the existing supply already has many features of an open access resource.

Overall, then, despite the formidable data problems and the difficulties involved, the model used here does suggest values in a range from very low to around $20 per hectare.

THE VALUE OF PLANT GENETIC RESOURCES FOR AGRICULTURE

Genetic and species diversity provide two important benefits to agriculture: a) the value of plant improvements and derived yield increases; b) a form of natural insurance against yield variability of homogenized systems. For conservation purposes, the issue is whether the value of these functions is maximized as the result of *in situ* genetic recombination, on farms and in the wild. If so, can mechanisms be devised for their capture? A related question concerns the distribution of benefits from the global adoption of new varieties originating in developing countries. The concept of `farmers rights', first adopted by the UN Food and Agricultural Organization as part of an International Understanding on Plant Genetic Resources, is implicit in the terms of the Convention on Biodiversity. The concept recognizes an historically unrewarded contribution to crop improvement and the need for a framework for corresponding compensation. Whether these largely retrospective claims will be addressed with any accuracy remains to be seen.

Measurement of the benefits of germ plasm diversity to crop development is notoriously difficult (Evenson, 1991). The genetic resources are seldom traded in markets and common landraces are often the product of generations of informal innovations based on wild species. Accession to collections held by international agricultural research centres subject selected landraces to further human intervention. Identifying the contribution of an original landrace to the success of a particular modern variety is therefore a complex task. Furthermore, the base materials used for breeding are themselves the result of a production function which includes labour and on-farm technology. Unravelling the returns to respective factors with any degree of precision, including a return to all historical intellectual inputs is improbable. Compensation, if feasible, may only be determined on a geographical basis.

Literature on genetic resource valuation rarely notes a distinction between the commonly quantified returns to plant breeding research and adoption (see for example Bohn and Byerlee, 1993; Mooney, 1993), and the more complex identification of the returns to genetic materials net of complementary inputs. Netting out human and technological contributions to agricultural production is complex, since an accurate picture of the contribution of genetic resources requires assessment of the net incremental yield value at *every* stage of recombination. As it happens, information on parentage and genealogy of many

common landraces is held by agricultural research centres worldwide. An accurate catalogue of yield effects of successive breeding stages and the necessary input cost information would require a much greater research investment. Cervigni (1993b) shows how a measure of the benefits of genetic material might be approximated in this way, using the difference between the benefits of an improved variety — commonly measured as the price multiplied by the yield increase — and the costs of all the other factors employed in breeding operations (capital labour etc). Data limitations mean that some degree of generalization of input cost information is inevitable. Costs and benefit streams arising in different periods imply some time discounting (see Annex at the end of this chapter).

Another method suggested by Evenson (1991) uses a methodology akin to hedonic pricing to relate yield value improvements to the genetic resources and other activities that were used to produce them. This approach may be particularly useful in revealing the relative contribution of genetic materials conserved *ex situ* to development of recent `successful' varieties. The incidence of success would also be indicative of the returns to wild species collections compared to developments based on existing genetic materials.

The complexity of modern and traditional breeding practices means that only the broadest approximation of plant genetic value is possible, and then only for the most common crops. This uncertainty is reflected in estimates of the contribution of South germ plasm to the value of food production in the North (via crop research centres like CIMMYT — The International Maize and Wheat Improvement Centre). For wheat and maize only, estimates range from US$75 million per annum for Australia, $500 million per annum for the US and $2.7 billion per annum for all OECD states (Mooney, 1993). It is not clear how much *ex situ* value-added is included in these estimates nor how they might be accurately assigned on a convenient per hectare basis to LDC agriculture. Such uncertainty should not be taken to imply that a vital global factor can remain unrewarded indefinitely.

GLOBAL VALUES: DEBT-FOR-NATURE SWAPS

Chapter 4 gave as examples of global values (a) the value of carbon storage in tropical forests and (b) the existence value attaching to biodiversity. This section looks at one possible way of uncovering the size of global existence values through `debt-for-nature' swaps. Debt-for-nature swaps (DFNs) involve the purchase, usually by an international conservation organization, but also by governments and even individuals, of developing countries' secondary debt in the secondary debt market. Such secondary debt is sold by existing holders at a discount, reflecting the market's judgement on the probability of repayment. In a DFN, the holder then offers to give up the debt holding in exchange for an undertaking by the debtor country government or an acting conservation organization to protect a given area, train conservationists, etc.

The idea of valuing the biodiversity so conserved through DFNs is that the payment made reflects some kind of willingness to pay on the part of the conservation body purchasing the debt. Different DFNs can be expected to come up with different implicit valuations since the nature of the `good' being bought will vary (eg the quality of the area protected will vary, and different packages of measures will be involved). Ruitenbeek (1992) has attempted to derive some WTP measures from six DFNs or deals which approximate DFNs. Here we outline the procedure used and show the implicit prices in a number of DFNs.

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