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Economics and the Conservation of Global Biological Diversity

Katrina Brown
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Working Paper
Number 2

UNEP

THE WORLD BANK

GEF Documentation

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This paper explores the relationship between economics and biodiversity conservation with minimal recourse to jargon, making the issue accessible even to the non-economist. It deals with the concepts of cost and benefit as they apply to biodiversity. Since biodiversity is an area where no clear measure of benefits exists, the process of project selection for a financial mechanism such as the Global Environment Facility (GEF) becomes especially complex.

The paper reviews concepts and measures of biodiversity, assesses what is currently known about extinction rates and species loss, and looks at efforts to place a value on biodiversity. It also investigates the question of why biodiversity is being eroded, and argues that the main economic causes of biodiversity erosion are the pressures exerted by population growth to convert available land, and under-investment in biodiversity. It recommends policies that reduce the returns on land conversion by eliminating subsidies and other economic policy distortions; and policies that aim to increase the returns on biodiversity conservation, for example, by conferring and enforcing property rights on those who sustainably manage resources, and by instituting mechanisms to capture the global benefits of sustainable land use. Finally, the paper considers ways in which the GEF might alleviate the problem of under-investment in biodiversity.

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Abbreviations

CITES	Convention on International Trade in Endangered Species of Flora and Fauna
CVM	Contingent valuation method
DNA	Deoxyribonucleic acid
ERR	Economic rate of return
GNP	Gross national product
GWP	Gross world product
NPV	Net present value
PCR	Polymerase chain reaction
TDR	Tradable development right
TEV	Total economic value
WTP	Willingness to pay

Data notes:

Dollars ($) are current US dollars unless otherwise indicated
1 hectare = 2.47 acres

Introduction

The Global Environment Facility (GEF) was established in 1990 for a Pilot Phase of three years. Its purpose is to channel investment and technical assistance funds to developing nations to assist them with their role in reducing four major global environmental problems: the depletion of the ozone layer, the threat of global warming, the degradation of international waters, and the loss of biological diversity. The GEF will continue into an operational phase, GEF II, in which it will consolidate its learning experience and disburse funds to the four focal areas, and toward solving problems of desertification and tropical deforestation insofar as they relate to the original four areas of concern. The modified GEF will be the interim funding instrument to achieve the goals of the two international conventions—the Framework Convention on Climate Change and the Convention on Biological Diversity—agreed at the Earth Summit in Rio de Janeiro in June 1992.

The disciplines needed for the efficient working of the GEF embrace science and social science. The need to think of global solutions and global costs and benefits has set a challenge for these disciplines, and the Pilot Phase of the GEF has been characterized by a productive questioning of the techniques and procedures whereby investments are usually appraised to see how they are best modified, if at all, in the global context. The reason for this reappraisal is that the GEF exists in the main to fund those investments which have limited or no rationale in terms of net gains to the host country, but which have ample rationale when considered in the context of global costs and benefits. Examples of the problems posed include:

- How best to choose between projects that are innovative but more costly compared with others that may be more traditional but yield more immediate, sizable reductions in local pollution
- How to measure the benefit of biodiversity conservation, and hence how to choose between competing conservation projects.

Because the GEF is charged in its operational phase with assuring the "cost-effectiveness of its activities in addressing the targeted global environmental issues" (GEF 1992, Principle IV), the measurement of cost and effectiveness is a focal concern. Cost-effectiveness has long been the concern of economics as a discipline, and the time seems right for some reflections on the way that economics can contribute to achieving the GEF's global objectives.

This paper is intended as a contribution to this process. It focuses very much on the perspective of the economist, but without (we hope) suggesting that economics is the only relevant discipline. It clearly is not. The paper also focuses only on biodiversity, for two reasons. First, the GEF has already commissioned work in the area of cost-effectiveness and greenhouse gas reduction (Mintzer 1992). The second is that biodiversity represents a major challenge to cost-effectiveness thinking: it will take a good deal more collaborative and interdisciplinary thinking to develop the kinds of

evaluative tools that are needed for this purpose. Rather, we have set out to say, broadly, what is known about the contribution of economics to biodiversity conservation, and to suggest some ways in which this "state of the art" impinges on GEF concerns. As such, we go beyond the immediate issues of measuring costs and effectiveness of conservation to investigate also why biodiversity is being eroded, for only by analyzing causes can we ultimately secure truly cost-effective policies.

Economists, like scientists, have a habit of using jargon. We have tried to minimize its use to make the paper comprehensible to the largest possible audience. In so doing we are conscious that the professional economist may feel we have oversimplified or failed to qualify statements sufficiently. We feel that the increase in communicability is a price worth paying for this oversimplification.

1 Concepts and Measures of Biological Diversity

The term biological diversity, often shortened to biodiversity, is commonly used to describe the number, variety, and variability of living organisms. 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 members of a species to regional ecosystems.

This chapter explains some of the key concepts of biodiversity, and approaches to the measurement of biodiversity and its components. Some estimates of rates of extinction are presented, and the problems in deriving such figures highlighted. The development of indicators to assess and monitor biodiversity that aid in the formulation of effective conservation strategies, are briefly described. 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 individual plants, animals and micro-organisms. Each species is the repository of an immense amount of genetic information. The number of genes ranges from about 1,000 in bacteria to more than 400,000 in many flowering plants. Each species consists of many organisms, and virtually no two members of the same species are genetically identical. Thus, even if an endangered species is saved from extinction, it probably has lost much of its internal diversity. Consequently, when populations expand again, they become more genetically uniform than their ancestors. For example, the bison herds of today do not have the same genetic diversity as the bison herds of the early eighteenth century (McClenagham et al. 1990).

Population geneticists have developed mathematical formulas to express a genetically effective population size. These explain the genetic effects on populations that have passed through the bottleneck of a small population size, such as the North American bison or African cheetah (World Conservation Monitoring Centre 1992). Subsequent inbreeding may result in reduced fertility, increased susceptibility to disease, and other negative effects that are termed "inbreeding depression." These effects 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 extremely slow, much variation can be lost. Conversely, "outbreeding depression" occurs when species become genetically differentiated across their range, and 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 deoxyribonucleic acid (DNA).

The significance of genetic diversity is often highlighted with reference to global agriculture and food security. This stresses the dependence of the majority of the world's human population on a few staple food species. These staple species have been improved by tapping genes from their wild relatives to foster new characteristics, for example, to improve resistance to pests and disease (Cooper et al. 1992).

Species diversity

Species are populations within which gene flows occur under natural conditions. Within a species, all normal individuals are capable of breeding with other individuals of the opposite sex, or at least 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 another species. Although this definition works well for many animal and plant species, it is more difficult to delineate species in populations where hybridization, self-fertilization or parthenogenesis occur. Scientists often disagree about where the necessary arbitrary divisions must be made.

New species may be established through polyploidy, the multiplication of the number of gene-bearing chromosomes. More commonly, new species result from geographic speciation, the process by which isolated populations diverge through evolution by being subjected to different environmental conditions. Over a long period, differences between populations may become great enough to reduce interbreeding. 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 level of classification. In ascending order, the main categories, or taxa, of living things are: species, genus, family, order, class, phylum, and kingdom. The exact number of species on earth is not known, *not even to the nearest order of magnitude*. Wilson (1988) estimates that the absolute number of species falls between 5 million to 30 million, although some scientists have put forward estimates of up to 50 million. At present, approximately 1.4 million living species have been described. The best catalogued groups are vertebrates and flowering plants. Such groups as lichens, bacteria, fungi, and roundworms are relatively under-researched. Likewise, some habitats are better researched than others. Coral reefs, the ocean floor, and tropical soils are not well studied. As we shall see, this lack of knowledge has considerable implications for the economics of biodiversity conservation.

The 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 several 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 extremely low species diversity. However, these areas also tend to be relatively small, which may be a factor that results in lower species numbers. In 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 to 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 comprises the size, number, 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 such ecological processes as productivity, hydrology, and soil generation. Nor does diversity neatly correlate with ecosystem stability, its resistance to disturbance, or its speed of recovery. There is also no simple relationship within any ecosystem between a change in its diversity and the resulting change in its component processes. On the one hand, 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. On the other hand, there can be cases where the converse is true. For example, if zebra and wildebeest are removed from the African savannah, net primary productivity of the ecosystem would decrease.

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

- The mix of species making up communities and ecosystems changes continually.
- Species diversity increases as environmental heterogeneity or the patchiness of a habitat does, but increasing patchiness does not necessarily result in increased species richness.
- Habitat patchiness influences not only the composition of species in an ecosystem, but also the interactions among species.
- 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.
- Both size and isolation of habitat patches can influence species richness, as can the extent of transition zones between habitats. These transitional zones, or "ecotones," support species that 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.
- 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.

Biodiversity is thus a complex and all embracing concept that can be analyzed and interpreted on many levels and scales.

Measurement of biodiversity

Biodiversity can be better understood when we examine exactly what we measure 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 on the continuing lack of a clear consensus about how biodiversity should be measured. Indeed, debates on measurement have comprised 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 indexes, of change in biodiversity.

Measurement of genetic diversity

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

- *Phenetic diversity.* This is based on measures of phenotypes, which are individuals that share the same characteristics. The 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 diffi-

cult 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 on a chromosome. This information is generally obtained using protein electrophoreses, which analyzes the migration of enzymes under the influence of an 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. Several different indexes and coefficients can be applied to the measurements to assess genetic distance (see Antonovic 1990). The advantage of the detection of allelic variation by electrophorese is 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 the selective importance of particular alleles.
- *Sequence variation.* This involves sequencing a portion of DNA using the polymerase chain reaction (PCR) technique. This technique means that only a minute amount of material, perhaps one cell, is required to obtain the DNA sequence data, so that only a drop of blood or a single hair is required as a sample. Closely related species may share 95 percent 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. Technically, however, 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. Alternatively, species richness might be 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 as a unit of biomass.

A more informative measure of diversity would also incorporate the "relatedness" of the species involved (Reid et al. 1992). Using a measure of taxonomic richness would imply that a region containing many closely related species would rank lower than one containing the equivalent number of distantly related species. However, as measures which could be applied to a range of different organisms have yet to be developed (but see Weitzman 1991a, 1991b), the richness of genera or families may provide a more accurate assessment of species diversity than simple measurements of species richness.

Measurement of community diversity

Many environmentalists and ecologists stress the importance of conservation of biodiversity at the community level. However, several factors make measurement and assessment of diversity at this level more nebulous and problematic. Many different "units" of diversity are involved at the supra-species level, including:

- Pattern of habitats in the community
- Relative abundance of species
- Age structure of populations
- Patterns of communities on the landscape
- Trophic structure
- 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 the ecosystem functions and services are also protected. These include pollutant cycling, nutrient cycling, climate control, as well as non-consumptive recreation, sci-

entific, and aesthetic values (see for example, Norton and Ulanowicz 1992).

Given the complexities of defining biodiversity at community and at ecosystem levels, a range of measurement approaches exist. As Reid et al. (1992) explain, many community attributes are components of biodiversity and might deserve monitoring for specific objectives. For example, generic measures of community-level diversity include biogeographical realms or provinces based on the distribution of species, and eco-regions or eco-zones 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, which are the more finely grained classifications that might be useful for policy-making. More policy-oriented measures include the definition of hotspots (based on the number of endemic species) and the delineation of megadiversity states. These concepts will be discussed in the context of using indicators for assessing and monitoring biodiversity.

Extinction

Speciation and extinction are natural processes, and Swanson (1992a) has described biodiversity as the net result of the processes of speciation and extinction. Species may be lost for a variety of reasons. Habitat loss and degradation are the most important causes of the present extinction crisis, but over-harvesting, the introduction of exotic species, and pollution also contribute. Global warming is expected to exacerbate the loss and degradation of biodiversity by increasing the rate of species extinction, by modifying the composition of habitats and ecosystems, and by altering their geographic areas (Peters and Lovejoy 1992).

Traditionally, from the Darwinian perspective, extinction is the fate of species that lose the struggle for survival. Taken to its logical conclusion, this view implies that extinction is a constructive process, eliminating obsolete species. It is now widely recognized, however, that this is not the case, since human intervention distorts the natural process. Many extinctions are non-constructive, and a species' ultimate demise is not a reflection of its "goodness" as a biological organism.

No precise estimate of the number of species being lost can be made, because the present number is not known. The vast majority of species is not monitored. However, there is no doubt that extinction is proceeding faster than it did before 1800. It seems likely that major episodes of species extinction have occurred throughout the past 250 million years, at average intervals of approximately 26 million years. According to Wilson (1988), the current reduction of diversity seems likely to approach that of the great natural catastrophes at the end of the Paleozoic and Mesozoic eras, the most extreme in the past 65 million years. Myers (1986) links present rates of tropical deforestation to a megadiversity spasm.

Table 1.1 Estimates of species extinction

Estimate of species loss	*% Global loss per decade*	*Method of estimation*	*Reference*
1 million species 1975–2000	4	Extrapolation of past exponentially increasing trend	Myers (1979)
15–20% of species 1980–2000	8–11	Species area curves	Lovejoy (1980)
25% of species 1985–2015	9	Loss of half species in area likely to be deforested by 2015	Raven (1988)
2–13% of species 1990–2015	1–5	Species area curves	Reid (1992)

Source: Reid (1992)

Table 1.1 shows some recent estimates of extinction rates. Many are based on estimates of habitat loss. The procedure estimates potential losses of species by extrapolation of rates of habitat destruction and calculation of associated extinctions using species area curves. It is based on principles of island biogeography, and recognizes a relationship between the number of species present and the area of a given habitat (MacArthur and Wilson 1967). Due to several problems associated with the use of this rather over-simplified equation for the calculation of extinction, figures calculated in this way might underestimate the expected extinction rate.

Assessing and monitoring biodiversity for conservation

For the assessment and monitoring of biodiversity as an aid to conservation policy, Noss et al. (1992) suggest a number of indicators. Useful indicators at the species level include monitoring of keystone species—those species of pivotal importance in their ecosystems upon which the diversity of the community as a whole strongly depends; and umbrella species (relatively wide-ranging species, such as large carnivores, whose protection would assure adequate amounts of habitat for many other species). Five categories of species have been used to justify special conservation efforts: ecological indicator, keystone, umbrella, flagship (charismatic), and vulnerable species.

At a community level, taxonomic groups that are relatively easy to monitor or that are particularly sensitive to environmental stress (for example, amphibians, fish, predatory birds, and butterflies) may be monitored for changes in abundance, species richness, or guild (a group of organisms that shares a common food source) proportions. Bibby et al. (1992) advocate the use of endemic bird species for identifying areas for conservation. At a landscape level, environmental changes such as changes in land use (say deforestation), human populations, demography, and even gross national product (GNP), may be used as indicators.

Increasingly, scientists argue that a species focus is not the best approach to the conservation of biodiversity. For example, Walker (1992) presents a functional approach to analyzing biodiversity. He argues that such a technique may be more appropriate for assessing conservation options than just a conventional taxonomic approach. This alternative approach focuses on the aspects of biodiversity that are critical for maintaining the resilience of ecosystems. Resilience is the capacity of an ecosystem to maintain its characteristic patterns and rates of processes such as primary productivity and nutrient cycling in response to variable environmental conditions. At the other extreme, Eiswerth and Haney (1992) argue that environmental economists and policy-makers tend to focus on the importance of species in isolation to one another, and on the number of species (species richness) in natural areas, to the exclusion of genetic and ecosystem diversity. They propose the use of estimates of inter-species genetic distance (originating in DNA-DNA hybridization) as a measure of genetic distinctiveness which should be considered in deciding conservation policy.

In biodiversity, as in other areas, economic and political factors rather than biological expedients often dictate the policies that are enacted. Soberon (1992) highlights the gaps between conservation theory and practice. For example, reserve sites are proposed for reasons of historical biogeography: richness of selected taxa, number and kinds of endemism, and other indexes that can be quantified. However, in practice, the size and shape of reserves are chosen as a result of political and economic considerations.

Conclusions

The design and implementation of conservation policies will depend on what we want to conserve, how we define biodiversity, and how we measure it in practice. It is clear that what we measure and how we choose to measure it affects our judgement and our ability to formulate and enact effective policy. From a conservation standpoint, it must be remembered that regions rich in some species are not necessarily rich in others. For example, in terms of species per unit area, Central America is more species rich than northern South America, but northern South America has more plant species. On the African continent, the species richness of butterflies is greater in west Africa just north of the equator, while the diversity of passerine birds, pri-

mates, and ungulates is greatest in central and east Africa, and plant diversity is greatest just north and south of the equator in west Africa. Mares (1992) has recently drawn attention to the incongruity in concentrating on Amazonia as the center of biodiversity. The drylands in South America are habitat to 53 percent more endemic mammalian species, and 440 percent more endemic genera than the Amazonian lowlands. On the basis of Mares' findings, if a single macro-habitat were chosen in which to preserve the greatest mammalian diversity in South America, it would be the largely continuous deserts, scrublands and grasslands. This is exactly the converse of the funding, research, and conservation strategies that have been employed to date.

The gaps in our knowledge of global biodiversity are significant, and basic work on inventories and systematics is still required. Noss et al. (1992) highlight the need for biodiversity inventories, which they visualize as a series of filters designed to capture elements of biodiversity at various levels of organization. At a national level, biodiversity inventories are best focused on the species, community (ecosystem), and landscape levels. While Erwin (1991) also acknowledges these gaps in knowledge, he highlights the potential pitfalls in attempting to estimate all the species on earth. Knowledge of the precise number of species might not be as important as recognizing the present rate of extinction due to human intervention, and devising policies to minimize it. This view implies that research efforts should be channeled into conservation and preservation of those species that we do have. Some scientists argue that this cannot be done effectively without more accurate data on existing species. Wilson (1988) has drawn attention to the declining number of taxonomists worldwide, the dwindling financial resources for their work, and the need to conserve this endangered species.

Biodiversity can be interpreted and measured on different levels of biological organization. Our knowledge is far from complete in many areas of genetics, in terms of total species and their distribution, and on ecological functions and processes. There is no scientific consensus on how best to measure biodiversity, but a number of indicators have been developed to inform conservation policy. Research is required on a number of fronts, including inventory, classification, mapping distribution, and monitoring. The implications for the economics of biodiversity conservation are potentially formidable. If we do not know what we are conserving and no reasonable consensus exists on how to measure biodiversity, how can effective policy be designed? While conservation policy may thus appear to be very much a hit-and-miss affair—and it does tend to be so for many reasons in addition to scientific uncertainty, as we shall see—it is important that policy goes in the right direction, even if an optimal policy is not apparent. However, before investigating what that direction is, the importance of biological diversity needs to be established. This may seem odd, but it is the failure to establish why biodiversity is important that explains why so much economics appears to proceed on the assumption that it is not important.

2 Placing a Value on Biological Diversity

Ethics and economics

Economists approach the issue of measuring importance in a particular way. The essence of their approach is that importance is measured by people's preferences. In turn, preferences are measured by looking at an individual's willingness to pay (WTP) for something. Economic value is then measured by the summation of many individuals' willingness-to-pay. So economic valuation in the environment context is about measuring the preferences of people for an environmental good (biodiversity) or against an environmental "bad" (loss of biodiversity). Valuation is therefore *of* preferences held *by* people. The valuation process is anthropocentric. The resulting valuations are in money terms because preference revelation is determined through people's WTP, or by inferring their WTP through other means. Moreover, the use of money as the measuring rod permits the comparison that is required between environmental values and development values. The latter are expressed in money terms, either in a dollar amount or an economic rate of return. Using other units to measure environmental values would not permit the comparison with development values.

The language of economic valuation is often misleading. Studies speak of valuing or pricing the environment. Similarly, changes in the environment affect health so it is necessary to find some valuations of changes in health status, with the ultimate change, of course, being the cessation of life itself. It is commonplace to find references to the value of life. Economists are apt to speak of the environment as a commodity, which leaves them open—perhaps justifiably—to charges that this is all the environment is worth. All these terminologies generate an unfortunate image of what economic valuation involves. What is being valued is not the "environment" or "life," but people's preferences for changes in the state of their environment, and their preferences for changes in the level of risk to their lives. There is no dispute that people have preferences for and against environmental change. There is no dispute that people are willing to pay to prevent or secure change: donations to conservation societies alone demonstrate this. The problem arises when this WTP is taken as *the* value of the environmental change. Many people believe that there are *intrinsic values* in environmental assets. They are of value in themselves and are not "of" human beings, values that exist not just because individual human beings have preferences for them. There is no reason to reject the idea of intrinsic values because the idea of measuring preferences is adopted. What is being assessed are two different things: the value of preferences of people for or against environmental change (economic values) and the value that intrinsically resides in environmental assets (intrinsic values).

Economic valuation is essentially about discovering the demand curve for environmental goods and services: the values of human beings for the envi-

ronment. This is another way of talking about finding willingness to pay.[1] The use of money as the measuring rod is a convenience: WTP happens to be one of the limited number of ways in which people express preferences. Once it is accepted that both forms of value exist, the issue becomes one of which values should inform and guide the process of making public choices. The answer is that since both values are legitimate, both are relevant to decision-making. Making decisions on the basis of economic values alone neither describes real world decision-making, nor would be appropriate given that governments and other agents involved in the development process have multiple goals. But one difference between the economic and intrinsic value approach is that economic values can, in principle, be measured. Intrinsic values cannot. If decision-makers do not feel the need for quantified assessments of gains and losses, then lack of quantification may not be an obstacle to decision-making. Otherwise it will often prove difficult to make choices between competing projects or alternative policies with differing environmental impacts.

The practical problem with economic valuation is one of deriving credible estimates of that value in contexts where there are either no apparent markets or very imperfect markets. If it is possible to derive such values, then it may well be that some measures of individuals' preferences will, in any event, capture at least part of what might be called intrinsic value. This will be so if the people expressing values for the environmental change in question themselves possess some concept of the intrinsic value of things. They may then be partly valuing "on behalf" of the environment as an entity in itself.

Many of the environmental assets that people generally feel are very important are in the developing world. Notable examples include the tropical rainforests, ecologically precious wetlands and mountain regions, and many of the world's endangered species. Many people feel these environmental assets have intrinsic value. They may express that view by speaking of the immorality of activities which degrade these resources, and of the "rights" to existence of trees and animal species. Bringing discussion of rights and intrinsic values into the policy dialogue can be counterproductive in contexts where the conflict is between conservation and, say, converting land to food production for immediate needs. If, on the other hand, conservation and the sustainable use of resources can be shown to be of *economic* value, then the dialogue between developer and conservationist may be viewed differently, not as one of necessary opposites, but of potential complements or alternative land uses that compete on an equal footing. The remaining stage rests on finding ways for the developing world to capture or appropriate the conservation benefits. If environmentalists in rich countries perceive value in conserving a rainforest in a poor country, this is of little consequence to the poor country unless there is a potential cash flow or technology transfer to be obtained. Economic valuation is therefore a two-part process in which it is necessary to:

- Demonstrate and measure the economic value of environmental assets—what we will call the *demonstration process*
- Find ways to capture the value—the *appropriation process*.

Total economic value

The economic value of environmental assets can be broken down into a set of component parts. This can be illustrated in the context of decisions about alternative land uses for a tropical forest, but the example can be generalized. According to a benefit-cost rule, decisions to "convert" a tropical forest, for say agricultural development, would have to be justified by showing that the net benefits from agriculture exceed the net benefits from conservation. Conservation could have two dimensions: preservation, which would be formally equivalent to outright non-use of the resource; and conservation, which would involve limited uses of the forest consistent with retention of natural forest. The definitions are necessarily imprecise. Some people would argue, for example, that ecotourism is not consistent with sustainable conservation, while others may say that it could be. Accepting the lack of precise lines of differentiation, the benefit-cost rule would be to convert the forest land only if the development

[1] Strictly, the demand curve traces out the willingness to pay for extra (or marginal) amounts of something. So the demand curve is a marginal willingness to pay schedule.

benefits minus the development costs are greater than the benefits of conservation minus the costs of conservation.

Typically, the benefits and costs accruing to the converted land use can be fairly readily calculated because there are attendant cash flows. Timber production, for example, tends to be for commercial markets and market prices are observable. Conservation benefits, on the other hand, are a mix of associated cash flows and non-market benefits. This fact imparts two biases. The first is that the components with associated cash flows are made to appear more "real" than those without such cash flows. There is a certain misplaced concreteness: decisions are likely to be biased in favor of the development option because conservation benefits are not readily calculable. The second bias follows from the first. Unless incentives are devised whereby the non-market benefits are internalized into the land-use choice mechanism, conservation benefits will automatically be downgraded. Those who stand to gain from, say, timber extraction or agricultural clearance, cannot consume the non-marketed benefits. This asymmetry of values imparts a considerable bias in favor of the land-use conversion option. As we shall see, these non-market benefits also have two spatial dimensions: benefits within the nation that possesses the resource, and benefits to other nations. Thus, the benefits of the tropical forest in nation A include such things as the watershed protection functions that the forest may have. The benefits to country B of A's forest includes the contribution that the forest makes to global climate stability, and the benefits reflected in B's willingness to pay to conserve the forest habitat because of its biodiversity. We shall refer to these different spatial benefits as domestic (or host country) benefits, and global benefits respectively.

Conservation benefits are included in total economic value (TEV). Total economic value for a tropical forest is explained in Table 2.1. This value comprises use and non-use values. Conservation is consistent with some sustainable uses of the forest, including sustainable timber harvesting.

Table 2.1 Total economic value and tropical forests

Total Economic Value =	*Use value* +		*Non-use value*
(1)	*(2)*	*(3)*	*(4)*
Direct value +	*Indirect value* +	*Option value* +	*Existence value*
Sustainable timber			
Non-timber products	Nutrient cycling	Future uses as per (1) + (2)	Forests as of intrinsic value, as a gift to others, as responsibility and stewardship
Recreation	Watershed protection		
Medicine	Air pollution reduction		
Plant genetics	Microclimate		
Education			
Human habitat			

Direct use values

Such values are fairly straightforward in concept but are not necessarily easy to measure in economic terms. Thus minor forest products output (such as nuts, rattan and latex) should be measurable from market and survey data, but the value of medicinal plants for the world at large is more difficult to measure, although estimates exist (see Pearce and Puroshothaman, 1992).

Indirect use values

These values correspond to the ecologist's concept of ecological functions. A tropical forest might help protect watersheds, for example, so that removing forest cover may result in water pollution, siltation, floods and droughts, depending on the alternative use to which that forest land is put. Similarly, tropical forests store carbon dioxide (CO_2). When they are burned for clearance much of the stored CO_2 is released into the atmosphere, contributing to greenhouse gas atmospheric warming. Tropical forests also store many species which in turn may have a wide range of ecological functions.

Option values

These relate to the amount that individuals would be willing to pay to conserve a tropical forest for possible future use. Option value is thus like an insurance premium to ensure the supply of something the availability of which would otherwise be uncertain. While there can be no presumption that option value is positive, it is likely to be so in a context where the resource is in demand for its environmental qualities and its supply is threatened by deforestation.

Existence value

This relates to valuations of the environmental asset that are unrelated either to current or optional use. Its intuitive basis is easy to understand because a great many people reveal their willingness to pay for the existence of environmental assets through wildlife and other environmental charities despite not taking part in the direct use of the wildlife through recreation. To some extent, this willingness to pay may represent "vicarious" consumption through wildlife videos and TV programs, but studies suggest that this is a weak explanation for existence value. Empirical measures of existence value, obtained through questionnaire approaches (the contingent valuation method), suggest that existence value can be a substantial component of total economic value. This finding is even more pronounced where the asset is unique, suggesting high potential existence values for unique ecosystems. Some analysts like to add bequest value as a separate category of economic value. Others regard it as part of existence value. In empirical terms they would be hard to differentiate.

Total economic value can be expressed as:

TEV = Direct use value + Indirect use value + Option value + Existence value

The usefulness of this classification is in practice debatable. Most contingent valuation studies distinguish use values from "non-use" values, but do not attempt to break down the component parts of non-use value (or "passive use" value, as some recent literature calls it—see Arrow et al. 1993). Others deny that existence value is relevant to economic valuation since it may be representing counter-preferential values based on moral concern, obligation, duty, altruism, and so on. But if we take the purpose of benefit measurement to be one of demonstrating economic value, however it may be 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 may be tempting to think that economists have captured all there is to know about economic value in the concept of TEV. But this is obviously incorrect. 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 (such as watershed protection). 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 for the importance of biological diversity. The first reason is based on the concept of economic value. If biodiversity is economically important we would expect this to show up in an expressed willingness to pay for its conservation. Shortly, we will show that this is indeed the case. The second reason is that 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. The third reason is that economic value does not capture—nor is it designed to capture—intrinsic value.

Global and domestic economic values in the GEF context

The distinction between domestic and global economic values is of fundamental importance for two reasons:

- The rationale for intervention by the Global Environment Facility is to capture the global values of conserving biodiversity, reducing greenhouse gas, preventing the pollution of international waters, and protecting the ozone layer
- Failure to appropriate global values of biodiversity conservation distorts the relative rates of return between conservation and "developmental" land use.

This section addresses the former issue, and the next section looks at the issue of the rate of return to conservation.

The GEF is primarily concerned with projects which, while not likely to yield net economic gains to the country in question, will yield net global benefits. These are termed Type II projects and are characterized by the following conditions:

Domestic costs (Cd) > Domestic benefits (Bd)
and
Global benefits (Bg) > Domestic costs (Cd)

The rationale for focusing on Type II projects as the prime focus for the GEF is straightforward. Type I projects are essentially development projects. If countries were not to achieve an excess of domestic benefits over costs, then investment would be inefficient and would not contribute to the developmental process. GEF is not part of official development assistance and is focused on global, not domestic, problems. Hence its concern is mainly with Type II, not Type I projects. Nonetheless, there will be circumstances in which some Type I projects will be eligible. This will be the case when global benefits (Bg) are judged to be large and the beneficiary could legitimately be expected to pay; and when recipient countries are clearly constrained by capital shortages.

The magnitude (Cd - Bd) is one interpretation, at its simplest level, of the concept of incremental cost. Acceptance of Type II projects thus requires that global benefits exceed incremental cost, or:

$$Bg > (Cd - Bd)$$

On rearrangement this rule is a simple cost-benefit rule, expressed as:

$$(Bg + Bd) > Cd$$

In general contexts, there are two ways in which the cost-benefit rule can be derived. The first is to estimate the monetary value of benefits. This means finding the willingness to pay of the host nation for biodiversity conservation (Bd), as well as the willingness to pay of the rest of the world for conservation—the global values (Bg). The second is to measure the effectiveness of the GEF intervention in terms of some index of the change in biodiversity that comes about because of the investment, typically the difference in some index of biodiversity with and without the project. The procedure then is to relate the cost of the project to the index to produce a cost-effectiveness measure.

There are many problems with the cost-effectiveness approach. Chapter 1 observed that there is no current

agreement on how to derive the relevant biodiversity indicators. Even if indicators can be found, there are problems in comparing different kinds of biodiversity, no matter what unit of account (such as ecosystem or species) is chosen. These are issues to be addressed by the GEF and others.

Is it possible to obtain monetary values for biodiversity conservation? The science of monetization has certainly advanced greatly and there is a significant body of work which bears on the issue (Pearce et al. 1992; Pearce 1993). Typically, what is being valued in these monetization exercises is not biodiversity per se, but usually some habitat such as a wetland or forest, or a particular species. The following sections consider examples of such economic values. In practice, some combination of indicators and monetized benefits is likely to be relevant to the evaluation of biodiversity conservation measures.

Domestic economic values

Biodiversity will tend to be conserved through habitat conservation. If it is possible to measure the domestic benefits of that conservation, then it is possible to construe those benefits as reflecting the benefits of biodiversity. This relationship between habitat values and biodiversity values is not very precise: it may be possible for some biodiversity to be reduced in a given habitat without economic values being impaired. But the values that arise from conserved and sustainably-used habitats will to some extent reflect the biodiversity in the habitat, since it is that diversity which contributes the economic value by providing, for example, a range of medicinal plants, a variety of minor products, or food.

Table 2.2, taken from Pearce et al. (1992), summarizes many of the studies of economic value of biodiversity as interpreted above. These values can be significant. Examples of market value of sustainable products are the estimate of some $6,800 per hectare (present value) for forest products in the Peruvian Amazon, and over $3,000 per hectare for local medicinal plants. An example of the non-market value is the $1,250 per hectare ecotourist values for Costa Rican forest, a value which accrues in the form of inferred willingness to pay over and above what is actually paid. Generalizing about these valuations is fraught with danger. We cannot, for example, extrapolate values of minor forest products to whole forests. Distance and limits of the market for the products will preclude this. The examples are illustrative rather than conclusive. But the evidence does suggest that local values are often higher than the price of land, or the net returns from "developmental" uses such as forestry and agriculture.[2]

Global economic values

Investigations into global economic values are comparatively few. They can be expected to increase as the demand for such valuations increases. In turn, this demand will increase as it becomes necessary to estimate the scale of international resource transfers under the various conventions agreed at the Earth Summit in Rio in 1992. However, early evidence already suggests that such values could be substantial. Various approaches to estimating global values are possible.

Contingent valuation

The first is to use the "contingent valuation method" (CVM) to find out what people in any one country are willing to pay to conserve biodiversity in another country. The CVM functions through sophisticated questionnaires which ask people their willingness to pay. Global CVMs of this kind do not yet exist.[3] Table 2.3 assembles the results of some CVMs in several countries. These report WTP for species and habitat conservation in the respondents' own country.

Debt-for-nature swaps

In a debt-for-nature swap an indebted country exchanges foreign debt for a newly created obligation, on which payments in domestic currency are used to fund an agreed conservation ("nature") program. The foreign debt is acquired at the substantial discount at which the debt is traded. Debt-servicing payments on the new domestic obligation are typically paid into a fund that finances the conservation activities. The price paid for the secondary debt can therefore be thought of as a willingness to pay to conserve the resource in question (Ruitenbeek 1992). Table 2.4 assembles available data on debt-for-nature swaps to see what the implicit price might be.

[2] The price of land should be related to the benefits of developing the land. More formally, the price of land is the present value of the flow of profits from the land. In practice, however, land prices often have a significant and speculative element.

[3] A global CVM is being carried out by the World Bank with respect to Madagascar's rainforests.

Table 2.2 Summary of studies on the economic value of biodiveristy

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: **Tropical forest** *Sources:* (1) Peters, Gentry and Mendlesohn (1989) (2) Gutierrez and Pearce (1992) (3) Ruitenbeek (1989a) (4) Mendelsohn and Tobias (1991) (5) Pearce (1991d) (6) Schneider et al. (1991) (7) Mendelsohn and Balick (1992) (8) Pearce (1990) (9) Watson (1988) (10) Kramer et al. (1992) (11) Guttierez and Pearce (1992) (12) Pinedo-Vasquez et al. (1992) (13) Solorzano and Guerrero (1988) (14) Schneider (1992)	(1) Sustainable harvesting in 1 hectare of Peruvian Amazon, (timber, fruit and latex 1987$). NPV hectare[1] $6,820 (local market values) relative to a net revenue $1,000 h[1] from clear-felling which risks uncertain regeneration, $3,184ha[1] plantations for timber and pulpwood or $2,960ha[1] from cattle ranching. (2) Estimated contribution of direct use to Brazilian GNP—$15b (3) Medicinal/genetic Net Present Value $7/ha over 126,000 ha (park area) or 426,000ha (with the additional buffer zone) This represents a minimum expected genetic value. Estimates depend on i) the probability of an area yielding a drug base ii) the method of valuation iii) an assumed extent of rent capture by local authority. Under certain assumptions the genetic/ medicinal NPV of tropical forest could be as high as $420 ha (See Appendix 1). (4) Travel cost valuation of tourist trips to Costa Rica's Monteverde Cloud forest. Average visitor valuation $35 (1988), producing a present value for trips assuming constant flows of $2.5m or extrapolating for foreign visitors, $12.5m. This gives a value per hectare in the reserve of $1,250 relative to the market price of local non-reserve land of $30–$100/ha. (7) Sustainable harvesting of medicinal plants in Belize *(local market values alone)* NPV $3,327per ha compared to $3,184 from plantation forestry with rotation felling. (9) Forest production (Malaysia) $2,455/ha compared with $217/ha from intensive agriculture. (3) Tourism value from the Korup $19/ha (10) Annual value of fuelwood to Malagasy households about $39 per annum	(3) Arising from sustained use of the Korup forest: Existence of Watershed functions affording protecton 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 (2010 represents the time horizon by which the continued use of the forest resources (in the absence of protection) would start to exhaust resources. The imputed benefit stream therefore represents the continued existence of resources. An imputed value of the expected loss from flooding resulting from alternative land use from 2010 on-wards: NPV of expected value of loss by 2040 is £1.6m ($2.84m) or $23 ha Soil fertility maintenance. Benefit imputed based on crop productivity decline from soil loss which would take effect from 2010 onwards (the without project scenario) NPV £532,000 ($958,000) or $8ha. (5) (6) Valuing Carbon sequestration; crediting standing forest with damage avoided from adverse climate change: $1.2b–$3.9b/year, depending on assumptions of: i) Damage estimate per tonne carbon estimated range $5–13 tonne. ii) amount released, itself dependant on assumptions of per hectare sequestration and annual deforestation rates. (8)(14) Carbon storage $1,300–5,700/ha/year (11) Total carbon storage value Brazilian Amazon $46b (13) Rio Macho Preserve, Costa Rica. Evaluates the replacement cost in terms of water services and energy generation resulting from reserve conversion to agricultural use.	Lower bound option value may be inferred from the current market value or foreign exchange earning potential of plant based pharma-ceuticals, (See Appendix 1) Attempts to gauge existence values in other contexts, rely on CVM to report WTP/willingness to accept compensation (WTA). To date only one study relating directly to tropical forests is available (10), although this does not report any foreign (explicitly non-use) WTP. However (2) set the existence value for the Brazilian Amazon at $30b, calcu-lated using an arbitrary WTP figure (observed from various CVM studies), aggregated across the OECD adult population. Donations to charitable funds may be one possibility to place CV evaluations in context; however, adichotomy exists between the observed reason for giving and actual use of funds. Problem of identifying organizations involved uniquely in forest protection. (3) Value of debt-for-nature swaps may provide an estimation of a WTP, reflecting a non-use value. Varying implicit valuation of different sites is reflected in the price paid by conservation bodies involved. Some swap transactions have aimed to preserve tropical forest ecosystems,(see Appendix 3). (10) Foreign visitor's WTP for the creation of the Mantadia National Park (1991). Bids ranged $75–$118 p.a., with sums being *additional* to existing prices paid. Multiplying these sums by the number of annual foreign visitor s to a neighbouring park (3,900) resulted in an annual WTP of $292,500–460,000, a PV of $3.64m–$5.73m (at 5% and 20 years) or $364–573/ha (10,000ha). These sums might represent use values as tourists were actually in the area.	(2)Brazilian Amazon: ($1991) Direct Use $15b Indirect $46b Existence $30b Total $91b/year NPV (using Krutilla & Fisher) $1,296b (10) CVM survey of villagers' WTA, to forego use benefits in the newly created Mantadia National Park (Madagascar). Implicitly their valuation will reflect a total economic value of the resource foregone. The survey revealed a per household sum of $13.91 per annum, which is aggregated over the affected number of households (347) to give $4,827 per annum PV (assuming payments for 20 years and discounted at 5%) $60,141 or $6/ha over 10,000 ha of park.	(1), Implicit ratios of 6.82, 2.14 or 2.3 depending on alternative use, but subject to qualifications regarding local elasticity of demand for harvested forest products. A similar exercise (12) in another area of Peruvian Amazon contradicts these estimates with a ratio of about 30 in favour of logging and rotation cropping. (2) Total present value $1296bn over 3.6b ha=$360/ha relative to a net revenue from clear felling of $1000/ha. The implied ratio of 0.36 will not be strictly representative since the calculation of Total economic value is not necessarily based on the assumption of sustainable use. (4) Implied for Costa Rica 12.5 which is the ratio of recreation value per hectare of protected area to the highest estimated price of land outside the park. (7) On the basis of local medicinal plant harvesting only, the implied ratio of 1.04 (9) Determination of market prices in this study is uncertain (ie world or local) implied ratio 11.3 (3) 1.07 total project ratio or 1.94 from the perspective of Cameroon when indirect project adjustments are included. These include figures for project related aid flows and value for uncaptured genetic and watershed values. (13) Implied ratio of 2

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 (Floodplains, Coastal wetlands, Wet meadows, Peatlands)** *Sources:* (1) Barbier, Adams and Kimmage (1991) (2) Semples et al. (1986) (3) Costanza et al. (1989) (4) Thomas et al. (1990) (5) Bergstrom et al. (1990) (6) Thibodeau and Ostro (1981) (7) Ruitenbeek (1991) (8) Hamilton and Snedaker (eds.) (1984) (9) Hanley and Craig (1991) (10) Van Diepen and Fiselier (1990) (11) Fiselier (1990a) (12) Danielson and Leitch (1986) (13) Turner and Brooke (1988) (14) Mcneely and Dobias (1991)	(1) NPV per acre ($1990) from the preservation of the Hadejia-Jama'are floodplain, Nigeria Agriculture $41 Fishing $15 Fuelwood $ 7 Discounted at 8% Other floodplain benefits: livestock and grazing non-timber forest products tourism, recreation, (including hunting), educational and scientific benefits (genetic and information value) (3) Louisiana. WTP PV at 8% ($1990) per acre. Commercial fishery $400 Fur trapping $190 Recreation $ 57 Storm protection $2,400 Total $3,047 (5) Louisiana. WTP PV at 8% ($1990) per acre Recreation $103 (6) Charles River, Massachusetts PV (1990$) per acre at 8%. Recreation $3400 Water supply $80,000 (8) Present Value per acre (at 8%) of Mangrove systems. Direct use from fisheries, forestry and recreation. Trinidad $15,000 Fiji $11,000 Puerto Rico $13,000	(1) Ground water recharge function for surrounding areas, potentially measurable by either WTP or using costs of ground water depletion on local agriculture—ie a *production function* approach—as a minimum benefit approximation. Other important functions: Flood Control and Storm Protection can in theory be approximated estimating alternative *preventative expenditure* or *replacement costs* for sea defences and dykes. In Malaysia the cost of rock escarpments to replace eroded mangrove fringe is typically around $300,000/km ($1990) (11). The same study quotes a 1987 E.C. estimate of the "inherent" value of mangrove protection to Guyana as $4bn, though there is no indication of how the figure is derived. Nutrient cycling will normally have a measurable effect on fishing and agricultural yields (in deltaic areas) the value of which might also be approximated by *replacement expenditures* on nutrients and compensating technologies. The value of wildlife habitats and life support functions will be reflected in the value placed on the continued existence of dependant species, (see under Existence values for some estimates) (14) Sustainable charcoal production from mangrove (Thailand) generates an annual national income of approx. $22.4m Net profits are nearly $4,000/ha for forests with average productivity of 230m^3/ha.	Significant option values fom future tourism, educational and scientific uses. existence values of wetland wildlife probably high although no explicit studies exist. (2) Some non-use values for wildlife (CVM estimates) 1990$/annum/person: brown bear, wolf, wolverine (Norway) 15 bald eagle (US) 12.4 emerald shiner 4.5 grizzly bear 18.5 bighorn sheep 8.6 whooping crane 1.2 blue whale 9.3 bottlenose dolphin 7.0 california sea otter 8.1 (9) Revealed WTP (CVM) for preservation benefits of blanket bog area in Scotland (1990) (once and-for-all payment) PV £164.68/ha (approx. $296.50/ha) implicitly representing the discounted future stream of user and non-user benefits. As such the value is interpreted as an option value. (See Smith [1987]) (12) An average annual amount ($343/acre) paid (by the US Fish and Wildfowl Service in 1980) to owners of wetlands in Massachusetts for *preservation easements*, can be taken to represent a minimum option Value for the ecosystem in an unaltered state. Similar conclusions could be inferred by looking at the average value of *management agreements* negotiated between conservation bodies and land owners in the UK. Such an *alternative cost approach* has revealed a value of £70/ha/per annum for coastal marsh and.	(7) Bintuni Bay mangrove ecosystem, Irian Jaya. NPV of whole system ($1991 discount rate 7.5%) $961–$1,495m, of which direct-use probably $152–534m. This value does not account for the high *cultural value* placed on the bay by the Irarutu tribe (10).	(1) Benefit/cost ratio expressed in terms the relative benefits accruing from alternative water use: $45 per 1,000m^3 of water maintained in the floodplain as opposed to *4 cents* per 1,000m^3 from diverted water. (4) From a similar analysis of the Ichkeul National Park, Tunisia, direct-use benefits amounted to $134 per 1,000m^3 compared to *negative returns* from diversionary use. Given the difficulty of generalizing with respect to alternative uses for wetland areas, informative cost-benefit ratios are difficult to provide. Where non-use values have been inferred from costs of imposing or agreeing land use constraints (the cost of which represent a discounted future benefit stream), the implicit cost-benefit ratio will normally be at least 1, because the compensatory payment from the recipient's perspective will have to be at least equal to the perceived opportunity cost.

Value category:	Direct use	Indirect-use	Non-use value option, quasi-option, bequest, existence	Total economic value	Benefit (sustainable use) /opportunity cost ratio
Ecosystem Type: **Rangelands (semi-arid) and wilderness areas** *Sources:* (1) Brown and Henry (1989) (2) Western and Thresher (1973) (3) Dobias (1988) (4) Child (1984)(1990) (5) Coulson (1991) (6) Dept. of National Parks, Zimbabwe (1991) (7) Jansen (1990) (8) Barnes (1990) (9) Imber (1991)	(1) Wildlife tourism. Viewing value of Elephants in Kenya $25m/per annum. The same study gives an indication of the extent of revenue forgone through sub-optimal park entrance pricing. A rough WTP survey revealed a potential consumer surplus as high as $25m/per annum (a sum almost 10 times the value of poached ivory exports and at least a 10% increase in actual expenditures). Since people were only asked their WTP to preserve elephants, consumer surplus for all wildlife viewing is presumably higher. (4) *Wildlife utilization:* Non-consumptive game viewing, lightly consumptive safari hunting and live animal trade, consumptive meat and hide production. **Zimbabwe:** illustrative examples: *Non-consumptive use:* Direct and indirect income accruing to the Matusadona National Park (1991) US$10.3m, 66% of which foreign currency (5). *Safari hunting:* Value for foreign visitors in 1990 was US$9m of which, trophies accounted for US$4m (6). *Consumptive value* Zimbabwe estimates it makes $4.7m/annum from the sale of elephant goods and services, a return of $75/km^2 over approx 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.	Indirect benefits from sustainable wildlife management: *Distribution of benefits to local communities* as a result of sustainable wildlife management schemes. (7) The Nyaminyami Wildlife Management Trust, Zimbabwe channeled approx Z$198,000 (1989) of wildlife revenues into local projects for health, housing, education and recreation. In addition the project was able to compensate local farmers for any damage incurred and offer cropped wildlife products for sale locally at subsidiized prices. Direct and indirect provision of employment. Improvements in local infrastructure and potential increases in land and property values. Significant saving in the hidden costs of land degradation and soil erosion arising from agricultural production in marginal areas. The role of elephants as keystone species diversifying savannah and forest ecosystems. Value added retained in the host country consists of net revenues accruing to: local airlines, tour operators, hotels, transport and cottage industries.	(3) Beneficial use project for Khao Yai National Park surveyed user WTP for continued existence of elephants at approx $7. Under certain assumptions of population and park use, the option and existence value of Khao Yai to Thai residents (for elephant preservation) may be as high as $4.7m/year . The extent of existence values might be approximated from the value of *vicarious tourism*—the consumption of books, films and TV programmes —particularly in developed countries, or from observed *charitable donations* to organizations involved in wildlife preservation. More crudely we might extrapolate on the basis of WTP information of visitors to wildlife sites in substitute countries like Kenya. In 1990 56% of overnight visitors to wildlife areas in Zimbabwe were foreign, of which 26% originated in Europe or North America (approx 151,000 visitors). Assuming 50% of these visitors reveal a similar WTP *in addition to* entry fees (in much the same way as in (1) i.e. a $100 permit for elephant preservation), extra revenue generated might amount to $7.5m per annum. (9) CV study preserve the Kakadu Conservation Zone (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–$1,520m , or a PV at 5% of between A$1m/ha and A$2.3m/ha over 5,000 ha. Both cultural and bequest values are likely to be significant in wildlife valuation although as yet few WTP studies reveal specific motivations.		(2) Ratio of wildlife tourism revenue per ha ($40) to income from extensive pastoralism ($0.80) 50. This ratio has probably increased significantly due to increasing value added in tourism. (4) Ratio of value of wildlife production (Z$4.20/ha) to Cattle Ranching (Z$3.58/ha in Zimbabwe 1.17. Calculation based on *economic rates of return* (as opposed to financial rates), and accounting for the relative environmental costs would in certain areas of the country produce ratios of between 2 and 5. (8) Provides PVs for returns from game viewing combined with some form of elephant cropping and for viewing alone in Botswana (1989). The ratio of the former to the latter range from 2.63 to 1.8 (depending on whether a 5 or 15 year horizon is considered) demonstrating the earning potential of consumptive uses. Comparison with the economic rate of return from cattle production on a per hectare basis could show ratios similar to those in Zimbabwe.

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) Carter et al. (1987) Hundloe (1990) (2) de Groot (1992) (3) Marcondes (1981) (4) Posner et al. (1981) (5) Schulze et al. (1983) (6) Hausman, Leonard, McFadden (1992)	(1) Estimating the socio-economic effect of the Crown of Thorns starfish on the Great Barrier Reef. A travel cost approach provided estimates of consumer surplus of A$117.5m /year for Australian visitors and A$26.7m/year for international visitors. The study showed that tourism to the reef is valued (in NPV terms) over and above current expenditure levels by more than $A1b. (2) Total direct use valued at $53/ha/year, comprising ($/ha/year): Recreational use 45 food/nutrition 0.7 Raw materials for construction 5.2 Energy resources 1.5 Ornamental resources 0.4 Biochemical and genetic resource values are also thought to be significant though no estimates are provided. Provision of employment directly or indirectly related to the National Park is a considerable benefit to the Galapagos economy (60% of 2,500 workforce). Tourism is the most important activity, contributing an estimated $26.8m to the local economy. (3) A form of Travel cost appraisal of the recreational value of the Cahuita National Park, 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 John, identified significant direct and indirect benefits associated with the park, particularly tourist expenditures 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 visitor's consumer surplus. Total benefit ($1980) approx. $8,295/ha over approx 2820ha of National Park on St. John. (6) Recreation demand study to value recreation use loss caused by the Valdez oil spill in Alaska; about $3.8m (1989)	(2) Estimates provided for the Galapagos National Park, Ecuador: $/ha/year Maintenance of biodiversity 4.9 Value of fish breeding (nursery function) 0.07 (applicable to 430,000 ha of marine zone). Watershed and erosion prevention functions 0.3 (applicable to terrestrial area of 720,000ha)	(2) Option value for the Galapagos National Park set arbitrarily at $120/ha/year which is the approximate sum of direct and indirect use values from the park. The uniqueness of the Galapagos ecosystem suggests that existence values are likely to be significant. (5) Describes a CV survey to value visibility improvements at the Grand Canyon (from reduced sulphur dioxide emissions). Mean bid ($1990/person/year) $27. A high level of familiarity may explain the high value respondents seem to have been willing to pay in this study (compared to bids for endangered species—see table 5.3). Higher WTP bids in habitat valuation studies have 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]).	(2) Total annual monetary returns from direct and indirect use approx $120/ha. In present value terms this represents $2,400/ha (at 5% discount rate) or almost $2.8b for the entire study area.	(3) Cahuita National Park ratio 9.54* (4) Ratio of total (direct and indirect) benefits to total cost 11.5* * A conventionally assessed ratio rather than one based on opportunity cost.

Source: Pearce, Moran and Fripp, 1992

Table 2.3 Preference valuations for endangered species and prized habitats

		(US 1990 $ p.a. per person)
Species		
Norway:	Brown bear, wolf and wolverine	15.0
USA:	Bald eagle	12.4
	Emerald shiner	4.5
	Grizzly bear	18.5
	Bighorn 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)
Other habitat		
USA:	Grand Canyon (visibility)	27.0
	Colorado wilderness	9.3-21.2
Australia:		
	Nadgee Nature Reserve NSW	28.1
	Kakadu Conservation Zone, NT[2]	40.0 (minor damage) 93.0 (major damage)
UK:	Nature reserves[3]	40.0 ('experts' only)
Norway:	Conservation of rivers against hydroelectric development	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 individuals only.

Sources: Norway—Dahle et al. (1987) (in Norwegian), Hervik et al. (1986); USA—Boyle and Bishop (1985), Brookshire et al. (1983), Stoll and Johnson (1984), Hageman (1985), Samples et al. (1986), Schulze et al. (1983), Walsh et al. (1984); Australia—Imber et al. (1991), Bennett (1982); United Kingdom—Willis and J.Benson (1988).

Implicit global willingness to pay in international transfers

Numerous debt-for-nature swaps have been agreed. The table below sets out the available information and computes the implicit prices. It is not possible to be precise with respect to the implicit prices since the swaps tend to cover not just protected areas but education and training as well. Moreover, each hectare of land does not secure the same degree of "protection" and the same area may be covered by different swaps. We have also arbitrarily chosen a ten-year horizon in order to compute present values, whereas the swaps in practice have variable levels of annual commitment.

Ignoring the outlier (Monteverde Cloud Forest, Costa Rica) the range of implicit values is from around 1 cent per hectare to just over $4 per hectare. Ruitenbeek (1992) secures a range of some 18 cents to $11 per hectare (ignoring Monteverde), but he has several different areas for some of the swaps, and also computes a present value of outlays for the swaps. But either range is very small compared to the opportunity costs of protected land, although if these implicit prices mean anything, they are capturing only part of the rich world's existence values for these assets. That is, the values reflect only part of the total economic value.

Finding a benchmark from such an analysis is hazardous, but something of the order of $5 per hectare may be appropriate.

Table 2.4 Implicit global willingness to pay (WTP) in international transfers: debt-for-nature swaps

Country	*Year*	*Payment (1990$)*	*Area m.ha PV*	*WTP/ha 1990$*	*Notes*
Bolivia	8/87	112,000	12.00	0.01	1
Ecuador	12/87	354,000 }	22.00	0.06	2
	4/89	1,068,750 }			
Costa Rica					
	2/88	918,000	1.15	0.80	3
	7/88	5,000,000			
4 parks	1/89	784,000			
	4/89	3,500,000	0.81	4.32	4
La Amistad	3/90	1,953,473	1.40	1.40	5
Monteverde	1/91	360,000	0.014	25.70	6
Dominican Republic	3/90	116,400			
Guatemala	10/91	75,000			
Jamaica	11/91	300,000			
Philippines	1/89	200,000 }	9.86	0.06	7
	8/90	438,750 }			
	2/92	5,000,000			
Madagascar	7/89	950,000 }	0.47	2.95	8
	8/90	445,891 }			
	1/91	59,377			9
Mexico	2/91	180,000			
Nigeria	7/91	64,788			
Zambia	8/89	454,000			10
Poland	1/90	11,500	unrelated to area purchase		
Nigeria	1989	1,060,000	1.84	0.58	11

Notes:
1. A discount rate of 6% is used, together with a time horizon of 10 years. The sum of discount factors for 10 years is then 7.36.

1. The Beni "park" is 334,000 acres and the surrounding buffer zones are some 3.7 million acres, making 1.63 million hectares (ha.) in all (1 hectare = 2.47 acres). 1.63 x 7.36 = 12 million hectares in present value (PV) terms.
2. Covers six areas: Cayembe Coca Reserve at 403,000 ha.; Cotacachi-Cayapas at 204,000 ha.; Sangay National Park at 370,000 ha.; Podocarpus National Park at 146,280 ha.; Cuyabeno Wildlife Reserve at 254,760 ha.; Yasuni National Park—no area stated; Galapagos National Park at 691,2000 ha.; Pasochoa near Quito at 800 ha. The total without Yasuni is therefore 2.07 m.ha. Inspection of maps suggests that Yasuni is about three times the area of Sangay, say 1 m.ha. This would make the grand total some 3 m.ha. The PV of this over ten years is then 22 m.ha. This is more than twice the comparable figure quoted in Ruitenbeek (1992).
3. Covers Corvocado at 41,788 ha.; Guanacaste at 110,000 ha.; Monteverde Cloud Forest at 3,600 ha., to give 156,600 ha. in all, or a PV of land area of 1.15 m.ha. Initially, $5.4 million at face value, purchased for $912,000, revalued here to 1990 prices.
4. Guanacaste at 110,000 ha., to give a PV of 0.81 m.ha.
5. La Amistad at 190,000 ha., to give a PV of 1.4 m.ha.
6. Monteverde Cloud Forest at 2023 ha. x 7.36 = 14,900 ha.
7. Area "protected" is 5,753 ha. of St. Paul Subterranean River National Park, and 1.33 m. ha. of El Nido National Marine Park. This gives a PV of land of 9.86 m.ha.
8. Focus on Adringitra and Marojejy reserves at 31,160 ha. and 60,150 ha. respectively. This gives a PV of 474,000 ha.
9. Covers four reserve areas: Zahamena, Midongy-Sud, Manongarivo and Namoroko.
10. Covers Kafue Flats and Bangweulu wetlands.
11. Oban Park, protecting 250,000 ha. or 1.84 m. ha. in PV terms. See Ruitenbeek (1992).

Carbon storage

Biodiversity conservation may often be an incidental effect, or "joint product" of some other environmentally beneficial activity. An example is carbon storage. All forests store carbon so that, if cleared for agriculture, there will be a release of CO_2 which will contribute to the accelerated greenhouse effect and therefore to global warming. In order to derive a value for the "carbon credit" that should be ascribed to a tropical forest, we need to know two things: the net carbon released when forests are converted to other uses, and 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_2 or charcoal and ash decays over time, releasing most of its carbon to the atmosphere within ten to twenty 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 a 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 will be attained after a relatively short time (forty-five years, according to German Bundestag, 1990). Table 2.5 summaries the carbon content of soils and biomass in the relevant land uses.

These data can be used to calculate the total changes in biomass and soil carbon as a result of land-use changes, as shown in Table 2.6. This table 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

Table 2.5 Carbon storage in different uses of tropical forests

	Biomass (tC/ha)	*Soils*	*Total*
Closed primary forest	167	116	283
Closed secondary forest	85-135	67-102	152-237
Open forest	68	47	115
Forest fallow (closed)	28-43	93	121-136
Forest fallow (open)	12-18	38	50-56
Shifting cultivation (year 1)	10-16	31-76	41-92
Shifting cultivation (year 2)	16-35	31-76	47-111
Permanent cultivation	5-10	51-60	56-70
Pasture	5	41-75	46-80

Source: compiled from German Bundestag (1990), and Houghton et al. (1987). Assumes carbon will reach minimum after 5 years in cropland, after 2 years in pasture.

Table 2.6 Changes in carbon with land-use conversion

	(tC/ha)			
	Original C	*Shifting Agric.*	*Permanent Agric.*	*Pasture*
Original C		79	63	63
Closed primary	283	-204	-220	-220
Closed secondary	194	-106	-152	-122
Open forest	115	-36	-52	-52

Note: Where range was given in Table 7 a mid-point is used here.

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

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.

The data in Table 2.6 then 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 tonnes to 200 tonnes of carbon per hectare.

We turn now to the value of this stored carbon. Nordhaus (1991a, 1991b), Cline (1992) and Fankhauser (1992) have produced provisional estimates of global warming damage. In the case of Nordhaus and Cline, the estimates are for the United States, and are extrapolated to the rest of the world. Fankhauser's analysis uses more worldwide information. Nordhaus' estimate suggests damage equal to some 0.25 percent of world GNP (GWP—gross world product), while Cline and Fankhauser suggest a figure of around 1.1 percent of GWP. Allowing for omitted categories of damage, Nordhaus suggests that 1 percent of GWP might be a central estimate, with 2 percent as an upper bound. Both estimates relate to a "two times CO_2 concentration" scenario—to damage done around 2030 and discounted back to the present. Taking the estimates produced with consistent discount rates, and taking 1 percent of Global World Product as the minimum damage, global warming damage would seem to be of the order of $7 to $18 per tC. If zero discount rates are applicable, then the upper range could be as high as $80 per tC. We use a "central" value of $10 per tonne carbon as the shadow price of carbon.

From Table 2.6 we can conclude that converting an open forest to agriculture or pasture would result in global warming damage of, say $300 to $500 per hectare; conversion of closed secondary forest would cause damage of $1,000 to $1,500 per hectare; and conversion of primary forest to agriculture would give rise to damage of about $2,000 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. Fearnside (1985, 1991) reports carbon loss rates of some 105 tonnes to 125 tonnes per hectare for conversion of primary forest to pasture, which are well below the

representative figures given in Table 2.6. But these apparently low emission figures suggest carbon credit values of $1,050 to $1,250 per hectare, three to four times the price of land in Rondonia.

The analysis strongly suggests that one of the cheapest options is to reduce tropical deforestation via some form of international transfer based on incentives not to burn the forests for clearance. Exactly how these incentives could be designed is a complex and separate issue, but there is scope for the Global Environment Facility to encourage such reductions in deforestation through its investment portfolio, and to experiment with tradeable burning rights.

Conclusions

Eliciting economic values for biodiversity is complex, but not impossible. The available evidence suggests that both domestic values (values to the host country) and global values may be very large relative to the conventional rates of return to land-use conversion for agriculture and even forestry. The two problems are: the rate of return to conversion land use is itself exaggerated by the presence of economic distortions such as subsidies; and a large part of the "rate of return" to biodiversity conservation accrues in non-marketed form and not to the farmer who considers converting the land. Domestic non-market values can be captured by suitable economic policies in the host country. But global non-market values can only be captured through some form of international resource transfer. This latter result is of course fundamental to the Global Environment Facility's purpose since the GEF can be viewed as a means of raising the host country's rate of return on conservation land uses. The importance of this will be underlined in chapter 3 where we explore the reasons why biodiversity disappears.

3 The Economic Causes of Biodiversity Erosion

In this chapter we focus on the economic causes of biodiversity loss. By economic cause we mean factors at work in the way that modern economies are organized and, perhaps more fundamentally, factors in the very evolution of modern economies. The basic proposition is that it is in the workings of the economy that we will find most of the factors explaining the decline not just in biodiversity, but in environmental capital generally, whether it is the ozone layer, the carbon cycle, tropical forests or coastal waters. Looking at economic causes does not mean we neglect more general factors such as population growth, for population growth is itself frequently to be explained in economic terms.

First it is necessary to look at the way economists have traditionally tried to explain the extinction process, or what we will call biodiversity erosion.

Extinction in the context of marine resources

The economic analysis of extinction was initially developed in the context of marine resources. This was the case because many of the earliest examples of modern species' endangerment occurred within that context. The Pacific fur seals suffered near extinction in the late nineteenth century due to over-exploitation. The blue whale experienced a severe decline in the same period. During the twentieth century the analysis focused on the decline of various fisheries, as the advance of technology made it possible to overfish entire stocks of various oceanic species.

Over-exploitation and extinction

In all of these cases it was *over-exploitation* that was the cause of the species' decline, since fishing and hunting pressures were occurring, with regard to many oceanic species, at levels that were unsustainable. Therefore the initial focus of the economic study of extinctions concerned the deleterious impacts of human hunting and fishing on various resources (Gordon 1954).

The study of over-exploitation was the first attempt at an economic analysis of the interface between human society and the remainder of the biological world. This resulted in the development of so-called bioeconomic models: models analyzing the interaction between human harvesting pressures and biological resource regeneration. The questions addressed in these models concerned the characteristics of a resource and resource management system that rendered them incompatible, so that the resource was incapable of sustaining the systemic pressures placed upon it by human society.

Economic analysis gave a short and simple answer to these questions. In the context of marine resources, exploitation of a resource was likely to be unsustainable if:

(a) The ratio of the price of the harvested resource to the cost of harvesting was "high;" and
(b) The natural growth rate of the resource was low" (Clark 1976, 1990).

These two conditions determined the ultimate impact of human harvesting pressures on a species. The intuition behind these conditions is simple. Condition (a) determines the incentives for human harvesting; a species with a high price relative to the cost of harvest is potentially very profitable, and therefore attracts substantial harvesting pressure. Condition (b) determines the capability of a species to sustain these pressures; a species with a low natural growth rate cannot regenerate itself at a rate sufficient to withstand substantial harvesting pressure.

Therefore the economic analysis of extinction in the context of marine resources is quite straightforward; the existence of financial incentives for significant harvesting pressures applied to slow-growing resources implies unsustainability. This explains the decline of such species as the great whale. Its high values as the provider of important oil products (in the nineteenth century) combined with its naturally slow growth rate meant that population declines were the likely result of the uncontrolled harvest, and this is the pattern that was in fact observed. Much the same could be thought to apply to land-based resources such as the African elephant and the rhinoceros. Both are comparatively easy to hunt, especially with the advent of high velocity rifles (low cost), and both have high value products (ivory and rhinoceros horn). We will soon discuss how far this transfer of extinction economics from sea to land is justified.

Open access as a cause of over-exploitation

In the marine context, however, the emphasis must be placed on the uncontrolled nature of all marine harvesting activity. Over-exploitation has often occurred in the context of the oceans precisely because the harvesting activities in this environment have been so poorly controlled. Until very recently, the oceans have been non-sovereign territory, and their resources have been subject to appropriation on a first-come-first-serve basis (Swanson 1991).

This form of resource management is known in economics as an open access regime. No one owns the resource and no one can effectively be prevented from making use of it. Such a regime installs a system of incentives based solely upon first appropriation, and this implies that no individual harvester has any incentive to discontinue harvesting the resource, because any of the resource that one harvester leaves behind will simply be captured by another.

In the context of an open access regime, extinction is a possibility if there are incentives to harvest the resource which exceed its capacity to replace itself. This is precisely what the bioeconomic models of extinction (in the context of marine resources) have demonstrated: open access regimes are a primary contributing force to extinctions.

Extinction in the context of terrestrial resources

As the elephant and rhinoceros examples suggested, it is tempting to apply the same bioeconomic models to land-based resources. But the economic analysis of extinction in the context of terrestrial resources is in fact very different from the analysis regarding marine resources. This is attributable to two fundamental differences, one on the societal side and one on the biological side of the bioeconomic model. These two features are:

- The existence of nation states in the terrestrial context
- Competition for land-based "niches."

"Optimal" open access regimes

First, there is no reason why the assumption of open access management should be carried over from the marine context to be applied to terrestrial resources, even to wildlife species. With regard to oceanic resources, the choice of the management system was not an option, because these resources lay in the international domain. However, on land there is a nation that has the designated responsibility for making management decisions with regard to the resources associated with any piece of territory; this is the economic meaning to be given to the legal concept of national territorial sovereignty, meaning that there are "owner-states."

If a terrestrial resource is managed by an open access form of regime, this is because some owner-state has chosen to apply this manner of manage-

ment to this resource. An owner-state may in fact choose to apply a management regime selected from a wide range of different forms of institutions to terrestrial resource management. For example, there are various forms of common resource management (such as communally managed pastures and forests), private property rights (exclusive to a single individual), and national ownership (such as national parks and forests). Any one of these resource management regimes may be the "best" in a given context and locality.

Paradoxically, even an open access regime can be an optimal choice (from an owner-state's perspective), even though it will always result in the inefficient over-exploitation of the resources subjected to it. This is because there are two potentially conflicting objectives at issue: the objective of maximizing the efficiency of the management of a particular resource (or group of resources), versus the objective of maximizing the efficiency of the management of the totality of a society's resources (natural, human and man-made). The efficient pursuit of the latter objective will imply the necessity of some trade-offs in regard to the pursuit of the former. Then it may be the case that inefficient forms of resource management are optimal from the owner-state's perspective.

In order to explain this concept clearly, it is necessary to understand that all state-level decisions concerning the regulation of natural resources are investment decisions, and the state is deciding implicitly whether the particular resource or region is worthy of an allocation of scarce societal investment funds. Efficient management of a given biological resource will require the regulation of harvesting activities with regard to a biological resource, in order to allow some amount of the existing stock of a species to remain for the purpose of generating future growth.

This capacity for growth is the reason that economists refer to a natural resource as natural capital. A natural resource is capable of generating a return just as with investments in any other form of asset. Investing in management regimes for a particular natural resource is the state's means for inducing investments in that resource. In other words, the crucial difference between the different forms of management regimes listed above is the aggregate amount of societal investment (by the group of harvesters) that results. For example, as described above, an open access regime creates incentives not to invest in the resources subjected to it; the harvesters of the resource will not view the resources as being worthy of investment because others will capture most of the benefits of their investments. On the other hand, if the state were to institute another institution (such as a property rights regime), then the harvesters would see some individual benefit to investing in the resources. In either case, the owner-state determines the ground-level investment incentives by its choice of management institutions for a particular region or resource.

Despite the fact that it has the power to choose the "efficient" management regime, it is sometimes optimal (from the state's perspective) to allow open access regimes to continue in place. This is because another important difference between these different management regimes (such as open access, common property, private property and national property) is the differential amounts of state resources that they require for their implementation. *In general, the creation and implementation of state institutions for protecting rights and monitoring production are costly affairs.* The one exception is that the institution of an open access regime costs the owner-state nothing (in the form of state spending requirements); other forms of state management regimes will require more significant commitments of state funding.

A zero-level of state spending with regard to the management of a particular resource or habitat may be optimal from a state's more general perspective because of the competing claims for national investments within a developing state. In short, it cannot be assumed that full and effective institutions will be warranted for the protection and production of all existing natural resources. The state must select its investments carefully, and allocate its scarce funds to those resources and regions that it believes will use them most productively.

Therefore, although an open access regime is the embodiment of an inefficient resource management

regime, it will be the optimal choice for many states in the context of severe scarcity of investment funds for institutional development. This implies that the fundamental cause of many land-based extinctions, especially those resulting from over-exploitation, is not the existence of open access institutions. Instead, *the fundamental cause is the existence of incentives in these developing states not to invest in the necessary management institutions in some regions of the country or in regard to some resources.*

Niche competition

The second important difference between oceanic and terrestrial resources is the extent of the niche available to the species. With regard to oceanic species, the size of the niche is exogenously determined; that is, the niche available to a species is determined by the carrying capacity of its natural environment. The species will be able to expand to the limits of its niche, as determined by its capacity to compete with other species (other fish, sea mammals, and so on) for the resources within that environment.

For terrestrial species, the amount of basic resources available to sustain a given species is no longer determined by a natural equilibrium of this sort, but by human choice. That is, the important difference between marine and terrestrial resources is the number of competing uses that humans have for their respective habitats. The main avenue by which humans interact with oceanic species is through harvesting and pollution; however, with regard to terrestrial species, the nature of the interaction is much more multi-faceted.

A terrestrial species will compete with humans (for the use of its habitat) in a multitude of different ways. Humans may consider making use of the habitat for purposes of agricultural production, and therefore the naturally-occurring species will have to be competitive with these in order to retain their lands. Alternatively, humans may consider using the land for purposes completely unrelated to the biosphere (such as residences or factories), and then the species must compete with these land uses as well.

The main habitat-displacing activity in the developing world is agricultural expansion, where agriculture includes both crops and livestock. Table 3.1 illustrates the changes in land area in selected countries. The table shows clearly that major increases have occurred in the areas of cropland in South America, Oceania and Africa, and significant increases in pasture-land have occurred in Central and South America. These broad aggregates conceal some major changes which are shown for selected separate countries.

With regard to these various forms of competition for land use, the danger to species is that they will be *undercut* rather than over-exploited. That is, these are resources that will lose their capacity to survive not because humans place too much pressure on their stocks but because humans convert their habitats to other uses. Part of the reason that such conversions take place is because the economic value of biodiversity is not transparent in the market place. This may be because there is insufficient

Table 3.1 Rate of conversion of land (late 1970s to late 1980s)
(percentage rate of increase)

Region	*Cropland*	*Pasture*	*Forest*
Africa	4.4	-0.5	-3.6
N and C America	1.1	3.1	1.0
S America	10.9	4.1	-4.6
Asia	0.8	-0.3	-5.3
Europe	-1.3	-4.0	1.1
ex-USSR	-0.2	-0.6	1.7
Oceania	11.6	-3.1	-0.6
Country			
B-Faso	27.5	0.0	-8.2
Côte d'Ivoire	20.8	0.0	24.1
Uganda	20.0	0.0	-8.1
Brazil	17.1	6.3	-4.2
Paraguay	46.7	32.6	-27.7
Suriname	53.4	11.1	-0.3
Bangladesh	1.5	0.0	-10.4
Malaysia	2.5	0.0	-11.0
Pakistan	3.3	0.0	17.3

Source: World Resources Institute (1992)

information about the benefits of biodiversity conservation (see chapter 2) or because there has been *under-investment* in conservation (see above).

Niche competition: statistical tests

If niche competition contributes to biodiversity loss we would expect this to be a testable proposition.

Table 3.2 Econometric studies of deforestation

A minus sign means that an increase in the variable leads to a decrease in deforestation. A plus sign means an increase leads to an increase in deforestation. Blank entries mean either not statistically significant or not tested for.

Deforestation significantly related to		*Rate of population growth*	*Population density*	*Income*	*Agricultural productivity*	*International indebtedness*
Allen and Barnes 1985		+				
Burgess 1992			+	–		
Burgess 1991	a)	–		+		+
	b)	–				+
Capistrano and Kiker 1990	a)			+		
	b)			+		–
	c)	+		+		–
Constantino and Ingram 1991			+	–	–	
Kahn and McDonald 1990		–				+
Katila 1992			+		+	
Kummer and Sham 1991			+			
Lugo, Schmidt and Braun 1981			+			
Panayotou and Sungsuwan 1989			+	–		
Palo, Mery and Salmi 1987			+			
Perrings 1992	a)	+	+	+		+
	b)	–	+			
Reis and Guzman 1992					+	
Rudel 1989		+		+		
Strafik 1992						
Southgate 1991		+			–	
Southgate, Sierra and Brown 1989		+				

Table 3.2 reports the results of various analyses of the statistical relationship between the destruction of habitat (a proxy for biodiversity loss), and demographic and economic variables such as population growth, population density, per capita GNP and international indebtedness. Appendix I shows the results in more detail.

The studies surveyed in Table 3.2 suggest, first, that there is no absolutely conclusive link between any of the selected variables and deforestation. However, cautious conclusions might be:

(a) The balance of evidence favors the niche competition hypothesis if that is expressed in terms of the influence of population growth on deforestation.
(b) Population density is clearly linked to deforestation rates.
(c) Income growth is fairly clearly linked to rates of deforestation, suggesting that deforestation has more to do with growth of incomes than with poverty—a result that runs counter to the popular interpretations of the causes of environmental degradation.
(d) The evidence on the role of agricultural productivity change is finely balanced. One would expect growth in productivity to lessen the pressure on colonization of forests (the coefficient of association should be negative). The two studies finding this association are for South America and Indonesia. The two studies finding the opposite association are for Thailand (Katila) and the Brazilian Amazon (Reis).
(e) The link between indebtedness and deforestation is also fairly clear. Perrings finds a positive link for tropical moist forests but not for other forests. Kahn finds a positive link, but Capistrano's models do not find such a link. Again, this ambivalence is at odds with the popular interpretations of the causes of environmental degradation.

Three routes to extinction

To summarize, there are three alternative routes to extinction for terrestrial species (as opposed to the single route for marine species). These are stock disinvestment, management resources diversion, and base resource conversion.

Stock disinvestment

As in the bioeconomic model applied to marine species, these are resources with high price/cost ratios but low growth. In that case, there are incentives to harvest the entirety of the resource (for its high value) and invest the funds in other assets (for their greater growth rates) (Clark 1973).

An example of this force in action is the deforestation of the tropical hardwood forests. These trees represent substantial amounts of standing value, but they have very low growth potential. It is economically rational to "cash in" the hardwoods and invest the returns in other, more productive assets so long as the economic value of conservation is either not known (the information problem) or is not appropriable (the "global benefit capture" problem of chapter 2).

Management resources diversion

These are resources of "medium" value but relatively low growth. Since they are slow-growing resources, they make little sense as assets; society has no incentive to invest in their growth capacity. In addition, on account of their relatively low value, they do not justify a commitment of substantial amounts of national resources for the management of the exploitation process. Then, the nation will allow these resources to be depleted through unmanaged exploitation.

Examples of this process include the depletion of many of the large land mammals, such as the African elephant. During the 1980s, sub-Saharan Africa lost half of its elephant population (from 1.3 million to 0.6 million). However, on closer inspection of the national population statistics, it appears that four countries alone (Sudan, the Central African Republic, Tanzania and Zambia) lost 600,000 thousand elephants between them. It is clear how these elephants were lost. These four countries fell at the bottom of the tables of African park and protection spending (averaging about $15 per square kilometer) (Swanson 1993). The decline of the African elephant in the 1980s was the result of these tacit open access regimes.

Base resource conversion
These are resources that are of little or no known individual value to humans. These biological resources are not over-exploited but undercut. They are lost because humans find alternative uses for the lands on which they rely. Those alternative uses reflect simple human need for food (the growing population problem) or the failure to appropriate the "true" economic value of conservation, or the existence of perverse economic incentives to convert land for biodiversity-friendly uses to uses inconsistent with biodiversity maintenance.

An example of this process is the depletion of many types of virtually unknown life forms when land is deforested and converted to other forms of use, such as cattle ranching. This branch of the force for extinction is generally termed the "biodiversity problem."

Regulating extinction: correcting investment incentives

The framework that these three forces imply focuses on the investment-worthiness of the resource. In essence, in the context of owner-states, all questions of extinction and species decline are based in the *incentives for under-investment*. The fundamental problem is that the owner-states do not invest in their diverse species (in terms of stocks, management and habitat). It is the failure of a state to provide for these requirements for a terrestrial species that inevitably results in its decline.

The decision to withhold these investments in regard to a given area of habitat implicitly derives from a determination that the naturally existing resources do not warrant the required investments. In regard to the developing tropical countries (where most diversity now exists), these are usually decisions not to invest in managing the forested frontier, thereby allowing wholesale over-exploitation of the diverse resources and encouraging widespread conversion to the traditional agricultural commodities (which are perceived as more productive).

For example, in the Amazon region, it is often the case that the owner-state refuses to engage in resource management on the frontier region before the conversions occur. Usually, the first vestiges of a management institution (for example, private property rights with some state enforcement) are put into place when the land is deforested, enclosed and converted to agriculture. This indicates the unwillingness of the state to invest in management regimes for the slate of diverse resources that naturally exist there (and the willingness to introduce these regimes with the introduction of different assets in those regions). Therefore the fundamental basis for any land-based species' decline is a state's determination, implicit or explicit, not to invest in that particular species (or, equally, its habitat). In order to regulate extinction, it will be necessary to operate through the perceptions of these states, affecting the determination of which resources are investment-worthy. Information about the economic value of biodiversity will help. It will also help to demonstrate that the relative economic rates of return to land conversion are distorted, for example, by subsidies, the removal of which would confer net benefits to the state in general. Procedures for capturing global economic value would also help restore a level playing field between conservation uses and land conversion uses. But the underlying force for making the conversions in the first place remains the need to produce food to meet growing human populations and, to some extent, the political requirement of establishing nationally loyal communities in border areas.

Niche appropriation and the specialized species
One very important reason for the continuing under-investment in diverse resources and diverse habitats is the bias towards investment in the specialized species. These are the domesticated (animal) and cultivated (plant) species that have been selected by humans to receive the vast majority of investment for purposes of meeting human consumption needs. They represent a minute proportion of the world's diversity, but they constitute the vast majority of the food consumed by humans. Only twenty species produce the vast majority of the world's food. The four carbohydrate crops (wheat, rice, maize and potatoes) feed more people than the next twenty-six crops combined (Wilson 1988).

The specialized species are of special importance because their mere existence indicates the nature of the underlying problem. These species are so prev-

alent because they are monopolizing the investments of human societies in biological production. In a world where human investments are now necessary for species survival, these *natural monopolies* in a handful of species imply non-investment in literally millions of others, and non-investment equates with extinction.

The extent to which this process of conversion underlies the process of extinction is indicated by the rates and locations of recent land-use conversions. During the twenty years 1960-1980, the whole of the developing world saw the proportion of its land area dedicated to the specialized species increase by 37.5 percent, while that same proportion remained constant in the developed world (where the conversion process is complete; see Repetto and Gillis 1988). At the "forest frontier"—countries where colonization of forest land is significant—these rates of conversion are even greater than the average, and continue to be so. For example, during the 1980s, Paraguay (72 percent), Niger (32 percent), Mongolia (32 percent) and Brazil (23 percent) have all experienced significant rates of conversion of lands to specialized crops; while Ecuador (62 percent), Costa Rica (34 percent), Thailand (32 percent) and the Philippines (26 percent) have all experienced significant conversions of lands to specialized livestock (World Resources Institute 1990).

Finally, it is important to note that the motivation for expanding the ranges of these few, specialized species is one of human niche expansion. These specialized species are mere instruments for the capture of photosynthetic product by human societies. The diverse biological communities are cleared, and these homogeneous ones installed, in order to convert the land to more beneficial production from the human perspective. The conversion of lands between biological assets (from diverse to specialized) is simply another manner in which humans benefit from adjusting their societal asset portfolio (Solow 1974).

An indicator that this has been a successful strategy for the human species is the phenomenal expansion of the human niche (as measured by the population of this species) since the introduction of agriculture. It is estimated that the human population on earth ten thousand years ago (before specialized agriculture) was approximately 10 million individuals. The advance of this particular technological frontier left much greater human population densities in its wake, first in the developed countries and now increasingly (with the Green Revolution) in the developing world.

The human population will reach 10 billion in the coming century. The movement to this much higher population level necessarily implies the capture of many other species' niches; the human species (one of five or ten million) now appropriates about 40 percent of all available photosynthetic product on the planet (Vitousek et al. 1984). This expansion has been occasioned largely through the instrument of conversions of land area to the use of the human-selected specialized species. It is the substitution of the specialized for the diverse on a global scale that is the most fundamental force for extinction.

Investment distortions

Nations under-invest in biodiversity because its rate of return appears to be less than that from alternative, competing uses of the land. But these rates of return to alternative land uses are themselves distorted: biodiversity uses have to be compared with two alternatives: (a) land uses in which inputs and outputs are "correctly" valued; and (b) the actual situation in which the value of inputs and outputs are distorted. These might be referred to as the level playing field comparison and the "unlevel" playing field comparison. Of course, this assumes that biodiversity uses are not themselves subject to subsidies and distortions. Typically, however, the bias is very much towards distortions in conventional land use.

Such distortions are widespread. Table 3.3 shows one set of subsidies to agriculture in selected countries for 1987. While some countries tax their agricultural sectors (these show up as negative numbers in Table 3.3) most subsidize agriculture. The subsidies shown, however, are recorded as sums received by farmers *in addition to the producer price.* What has to be remembered is that the producer price may itself be artificial in that it is controlled by government. As an example, rice and cereal prices paid to farmers in Japan were six to

seven times the border price. In addition, farmers received subsidies in excess of 100 percent of the producer price. Table 3.3 shows that the countries which subsidize over and above producer prices tend to be the developed economies of Canada, the European Community, Japan and the United States, together with the newly industrializing countries of South Korea and Taiwan. But subsidies are pervasive, as the table shows.

Table 3.4 shows the magnitude of distortion arising from the level of producer prices as well as the subsidies granted (shown in Table 3.3). Only a few selected countries are shown. Once again, the data are rudimentary but instructive. In Bangladesh, for example, the subsidy structure to wheat makes little difference to what producers would get if border prices ruled. The same is true for Indonesian rice. In China, the effect of the subsidy structure is in fact to tax farmers so that Chinese producer incomes per hectare are actually below border prices. But in Japan (wheat) and South Korea (rice) it can be seen that not only is the producer price well above the border price (seven and four times respectively, but further subsidies of around 100 percent are paid in addition to the border price. The effect is to make the returns to a hectare of wheat production in Japan around fourteen times what it would be without subsidies and without adminis-

Table 3.3 Producer subsidies as a percentage of producer prices in agriculture 1987: selected countries
(%)

	Rice	*Corn*	*Sorgh.*	*Soya*	*Wheat*	*Beef*	*Sugar*	*Other*
Argentina		-23	-42	-10	6			
Australia					4	4	16	
Bangladesh	35				-1			
Brazil	95	5		-9	59	1		
Canada		37		22	51	10	54	
Chile		20			44		50	
Colombia				63			21	coffee -35
Egypt	-126	15			4		45	
EC	56	56		46	50	40	41	
India	4	4	-10	11	7			
Indonesia	8							
Japan	97			59	108	72	70	
Kenya	-7	13						coffee -31 tea -41
Mexico		75	59	52	20		18	cotton 60
Nigeria	49	6			64			cotton 83
Pakistan	-1 and -128*				-18	-17	11	
Poland					54	36	29	
South Korea	84	77		85		47		
Taiwan	46	82	83	72	79	22	28	
Thailand	5							
Turkey		19			42	4		
USA	49	46	43	8	63	10	60	

Source: Adapted from Webb, Lopez and Penn (1990). Note that these subsidies are in addition to producer prices which may themselves be higher than border prices — see Table 3.4.
** Different types of rice.*

Table 3.4 Two sources of subsidy to agriculture: selected countries 1987

	(1) Yield t/ha	(2) Border price $/t	(3) Producer price $/t	(4) Subsidy % of PP	(5) Producer price $/ha (1)x(3)	(6) PP+Sub $/ha	(7) Revenue/ Border price (6)/(1)x(2)
Wheat							
Bangladesh	2.5	164	172	-1	426	421	1.03
China	4.1	151	130	-17	527	458	0.74
Japan	5.7	184	1201	108	6801	14147	13.6
Nigeria	1.1	143	323	64	362	594	3.7
USA	4.3	70	96	63	416	679	2.2
Rice							
Indonesia	3.7	192	190	8	706	763	1.1
S. Korea	5.9	257	1095	84	6499	11958	7.8

Source: Authors' calculations.

tered producer prices, and around eight times in South Korea. Clearly, with distortions at this level, there is nothing remotely like a level playing field for biodiversity uses.

The failure to capture global economic value

The rate of return to biodiversity conservation is further distorted by what economists call "missing markets." Chapter 2 indicated two such markets that are highly relevant to biodiversity: carbon storage in tropical forests and the existence value possessed by individuals in one country for wildlife and habitat in other countries. The carbon storage values are potentially very large, assuming the science of global warming is fulfilled. Tropical forest "carbon credits" could be of the order of $1,000 to $2,500 per hectare, dwarfing the investment returns from conventional land-use options such as agriculture. Comparisons with forestry values also suggest high conservation values, where conservation is taken to mean sustainable utilization. Rates of return to unsustainable forestry could be as high as $2,000 per hectare (Pearce et al. 1992), but figures in the range of $1,000 to $2,000 are more likely. Rates of return to sustainable forestry are perhaps $230 to $850 for Malaysia (Vincent 1990) depending on assumptions about yield, and the discount rate.

In the language of chapter 2, if host countries could capture the total economic value of habitat conservation, and if the resulting cash flows could be directed towards those who make decisions about land use, then, clearly, the relative rates of return to conservation and conversion would change in favor of conservation. Nation states would then have less incentive to under-invest in biodiversity.

Conclusions on the fundamental forces underlying extinction

Extinction in the marine context may readily be explained by the existence of open access regimes, and by the relative profitability of harvesting sea species relative to the alternative uses of capital. But open access in the marine environment arises because nations have not, until recently (but still incompletely) exercised national property rights over the oceans.

Extinction in the terrestrial context cannot be explained in the same way. It is necessary to ask why nations allow natural resources to be depleted to the point of extinction. Since they are the "owners" of the land, the process must reflect some choice not to invest in those resources. This process of under-investment has to be seen against the backdrop of the need to feed growing populations and, occasion-

ally, political objectives with respect to the settlement of land. But the *rate* of land conversion is accelerated further by (a) lack of information about the economic value of conservation; (b) deliberate policies which encourage land conversion—such as subsidies; and (c) "missing markets"—the inability of nation states to capture the global economic benefits of conservation. It is arguable then that, while niche competition remains a fundamental cause of biodiversity loss, the rate of conversion would be slowed if the information, distortionary policy, and missing market problems were resolved. Moreover, as the statistical analysis showed, there are other forces at work as well.

What relevance has such a view for the Global Environment Facility? First, it helps to put the GEF activity in context. GEF was not established to *resolve* the problem of biodiversity loss, but to contribute to a slowing down process. If one main force explaining the losses is population growth, then GEF can expect to have a limited role to play in conservation since it has no powers nor any remit to change that fundamental process. But GEF does address the second issue of under-investment. Essentially, GEF's funding of the incremental cost of projects (see chapter 2) raises the rate of return to conservation projects in the host country. And it does this by focusing on one of the "failures" in the way the competition between conservation and land conversion works—the failure to appropriate global benefit. Nor should the low level of funding of GEF be too disadvantageous provided it can use its limited funds to lever private sector capital, an issue discussed briefly in chapter 5.

4 Reversing the Decline

From the discussion of the nature of the extinction problem in chapter three, it is apparent that there are two different spheres to be considered, marine and terrestrial. These spheres are distinct biologically, economically and (most important) institutionally.

Regulation of oceanic resources

In the oceanic context there is no owner-state with the designated responsibility for management of a given marine resource. The regulation problem concerns the construction of an owner-manager regime with regard to the resources which are, in fact, international resources, falling outside any one state's territorial jurisdiction. The role of regulation is to develop a management regime that would mirror that constructed by an owner-state, if one existed.

The development of an international resource management regime is, in theory, a simple task. It concerns the performance of a straightforward four-step process in regard to the resource:

(a) Assessment of stock level and growth capacity of the resource;
(b) Determination of the aggregate optimal off-take (quota) for the resource;
(c) Allocation of the overall quota to individual states; and
(d) Enforcement of the individual state quotas.

Precisely this approach has been attempted for about half of this century by the various fisheries commissions of the world. It was also the initial rationale underlying the International Whaling Convention. Most of these international treaties functioned through the creation of a commission whose task it was to develop a scheme similar to that outlined in (a) - (d).

In practice none of these commissions has been a success in terms of the management of its resources. This result has been occasioned not by a failure to appreciate the nature of the management solution to the problem, but rather by a failure of the parties to accept and implement the managed solution.

Instead, the various states have over-depleted oceanic resources by failing to agree how to distribute the gains from implementing the managed solution; these are *multilateral bargaining problems*. When each state argues that it should receive a larger share of that gain, pressure is created to revise the aggregate off-take in order to accommodate all the demands. These individual demands are then met through: increased aggregate quotas; state withdrawal from the international commission (reservations and non-accession); and/or state non-enforcement of its individual quota. The ultimate effect is the inefficient international management of the resource. The existence of these international bargaining problems has resulted in the over-depletion of many important oceanic resources.

Regulation of land resources

Although the problem of extinction on land is very different in character from that in the context of oceanic resources, the policy approach applied on

land derives directly from the analysis of the problem developed for the sea. This has led to a largely ineffective and necessarily inefficient set of endangered species policies, as traditionally defined.

The leading piece of legislation regarding endangered species is the Convention on International Trade in Endangered Species of Flora and Fauna. Its policy is to focus on the identification of endangered species and the withdrawal of the demand for these species or their products, and the criminalization of their supply. Endangered species policies developing out of this convention therefore operate through a *system of bans*. An importing state is required to ban all imports of products derived from a listed endangered species, and an exporting state is required to ban all exports of products derived from such a species.

This system of bans is required of all parties to CITES (now more than 125 nations). In addition, there is the requirement that the member states adopt internal legislation implementing the terms of the convention. Many states, particularly developing countries, have implemented absolute bans on all wildlife exploitation. In many developing countries it is illegal to hunt, capture, trade or export any part of the wildlife resource. This is true for most of the states of South and Central America. For example, Brazil and Bolivia have total bans on all wildlife exports, as does Mexico. Many of their neighbors have partial or full bans in place. In sub-Saharan Africa, there are a half dozen states with complete wildlife exploitation bans in place, while many others have severe use restrictions.

This approach to endangered species policy is based on the *over-exploitation theory of extinction* described in chapter 3. It operates indirectly through the economic system, by lowering the unit price of a species' products (through the destruction of demand) and by increasing the cost of supply (through criminalizing the production process). The narrowing of the price-cost ratio would have the effect of reducing the pressures on a resource within an open access regulatory framework, if the fundamental cause of its decline is over-exploitation.

However, as discussed in chapter 3, the incentive to over-exploit stocks of biological resources is one of only three avenues to extinction for terrestrial resources. The other two, the competition for land and management resources, are equally, or more commonly, the incentives leading to extinction. Therefore a system of bans would be an apt policy for over-exploited oceanic species and the slowest growing (and highly valuable) terrestrial species, but the policy would have no positive impact on the incentives resulting in the extinction of other terrestrial resources. Species that are in decline on account of competition for lands (general biodiversity), or competition for state investments in management (such as elephants), will not be assisted by the prevailing set of endangered species policies.

In fact, there is an argument to be made that the existing policies entirely misapprehend the core of the extinction problem—the creation of incentives for owner-states to invest in (rather than convert) their remaining diverse resources. A system of bans, and the resulting reductions in the profitability of all diverse resources, is the antithesis of a constructive approach to the fundamental problem of extinction.

International regulation of extinction—creating incentives for owner-states

The objective of international extinction policies should be the inducement of investments by owner-states in their diverse resources. This solution is suggested by the nature of the problem, as developed in chapter 3. In essence, if an owner-state does not view its diverse resources as worthy of significant investments, then it will be optimal (from that state's perspective) to allow a continuation of the conversion of its lands to more specialized uses. Such conversions can occur through the managed "mining" of existing resources (as with hardwood forests), via the unmanaged over-exploitation of existing natural resources (as with the African elephant), or by the removal and replacement of the diverse with the specialized (as with land-use conversions to cattle and crops). All three routes to extinction have the same ultimate result (changed land use) and the same underlying cause (perceived investment worth). The objective of international extinction policy, within this framework, must then be to alter the perceived relative investment worth of diverse resources within individual owner-states.

Global versus local optimum

The global problem of extinction comes down to a basic divergence between what investment in diverse resources is desirable from a global point of view and what is desirable from a local point of view. The owner-state responds only to the perceived national (internal) relative advantages of diverse versus specialized resources. These are the same incentives that are found in every state, and that have resulted in the almost complete conversion of the lands of most of the developed states in the world. For example, the amount of unaltered habitat of at least 400 square kilometers is zero in Europe against a global average of 30 percent (World Resources Institute 1990).

The difference between the previously and currently converting states is not so much a matter of differences between what is locally desirable, but rather the change in what is optimal globally given the state of the global stocks of diverse resources. In essence, the cost of each land conversion (from diverse to specialized) is not the same from the global perspective because there is a *global stock effect*. As stocks of diverse resources are replaced from the same roster of specialized resources across the globe, the range of global diversity continues to narrow. The initial narrowing of this roster probably had little consequence for global biological production; however, the final conversions will have a very substantial impact. A world constituted of none other than the relative handful of cultivated and domesticated species would not support a sustainable production system for human consumption needs.

It is this *uninternalized increasing costliness of conversion* that is the core of the global biodiversity problem. The individual state considering land-use conversions does not consider the effects of its actions on the global stocks of diverse resources, because there are no benefits from doing so. Therefore the unregulated global conversion process may continue well past the global optimum as individual states implement only locally optimal conversion policies.

International extinction policy—investing in the internalization of the global stock effect

The best policy for regulating the conversion decisions of individual owner-states is the creation of systems that cause these states to consider the external effects of their decisions. In economic terms, the objective is the *internalization of the global stock effect* of diverse resources in the owner-state's decision-making framework. The rationale is that if an owner-state considers the global benefits rendered by diverse resources when making its conversion decision, it will only decide in favor of conversion when that is globally optimal. This internalization of global externalities has the effect of making the perceived local optimum coincide with the global optimum.

Such a policy implies that international regulation needs to be directed to the creation and maintenance of a *global premium* to investments in diverse resources. This is an additional return, created through funding by the international community, that will flow to owner-states investing in their diverse resources. There are two logically distinct approaches to the creation of such premiums: international subsidy agreements and market regulation agreements. However, there are a number of different forms (international parks, international management subsidies, intellectual property rights, producer cooperative agreements, consumer purchasing agreements and resource exchanges) that either of these approaches may take.

International subsidy agreements

Although it is under-investment that generally drives extinction, it is the specific costliness of particular resource requirements that is the proximate cause of extinction. Extinction is caused most directly by the refusal to allocate scarce societal resources to the lands or the management that diverse resources require. It is the refusal to purchase these life-sustaining factors for certain forms of biological resources that is the direct cause of species decline.

These sources of decline can be remedied through a system of strategic international payments. This system of payments would necessarily be conditional upon the owner-state's application of them to the purchase of the required factors for specified diverse resources. The payments would thus be restricted to use for the purchase of land and management for a particular resource or region.

Such a system of payments is constructed in a manner that constitutes a self-enforcing interna-

tional agreement. The voluntary cooperation of countries in an international agreement means that national sovereignty is respected rather than infringed. So the first principle for constructing an effective system for regulating biological diversity is to recognize terrestrial biological resources as national resources. Once this point is recognized, the approach to registering global preferences in the regulation of national resources is clear; it requires the inducement of changed national policies via strategic payments. In this way, host-states are induced to exercise their national sovereignty rights in globally-preferred ways, but only because it is in their own perceived best interests to do so.

A crucial feature of any international scheme of payments based on conditionality (here, payment conditional on specific application) is its necessarily dynamic nature. It is only possible to restructure the owner-state's decision-making process if the payment is offered at the end of each period in which the state takes the specified action. The payment is the *global premium*, supplementing the state's return, received for pursuing the global rather than the local optimum.

A second important feature of this system of payments is assurance. As emphasized in the previous sections, investments in diverse resources represent investments in specific forms of assets. It is the expectation of a future flow of benefits from these investments that will ultimately induce them. Therefore the global community must, in order to alter host-state investment patterns, itself invest in two distinct ways: (a) the creation of a stream of enhanced benefits for host-states investing in diverse resources; and (b) the creation of assurance that this stream of payments will flow to investing states in perpetuity.

This latter object is equally important because, without assurance, the enhanced benefits are meaningless; states will not alter their long-term investment behavior without assurance that these benefits are non-discretionary. The choice of an investment path for a given state's development is not redirected by means of one-off impermanent injections of funds. This is the reason that global investments in biodiversity must take the form of institution building. The permanent redirection of development paths will require the creation of institutions for that purpose. The suggested reforms outlined in this paper should be interpreted as potential directions for institution building of this nature.

An example of such an international factor-subsidy scheme might be an *international parks agreement.* Such a program would in effect buy the use of the land for the diverse resources that exist there. The role of the international community would be to pay the rental price of the land each year that the national park remained unconverted. In order to manage the park, it would probably be necessary for the international community to provide a subsidy for management services as well. If the international community wished to maintain diverse resources at the lowest levels of exploitation (for non-consumptive activities such as tourism and filming), then it would be necessary to provide almost complete subsidies for the foregone land development opportunities and management services required.

However, the maintenance of a stock of diverse resources will often be compatible with a wide range of forms of resource utilization, other than those that are of the lowest intensity. In that case the international subsidies may be reduced in accordance with the extent to which development opportunities are allowed in the region. This, in essence, is the *development rights* approach to diverse resource conservation; to the extent that the international community wishes to reduce the development intensity away from the local optimum (of high intensity conversion and use), it must be willing to provide a stream of ex post payments to compensate for the foregone development.

Another example of an international subsidy scheme would be a *resource franchise* agreement. This would be a three-way agreement (between owner-state, international community and franchisee) in which the international community would provide a stream of rental payments to the owner-state in return for its agreement to restrict use of a given piece of land to uses specified within the franchise agreement. The land would thus be designated for use, but only for limited uses amounting to much less than complete conversion (for example, for

extractive industries such as rubber-tapping, and plant and wildlife harvesting). The land would then be franchised to an entity that could use it only for purposes compatible with the franchise. If the state failed to enforce the franchise agreement, it would forfeit its annual rental fee.

The benefit of a franchise agreement over an international parks scheme is that it provides a stream of benefits to fund the management of the operation. Under the franchise agreement, it is the responsibility of the franchisee to provide management services from the returns it generates in its operation of the franchise. In this fashion, the international community is able to "contract out" the provision of management services, while (through the rental fee and restricted use clauses in the franchise agreement) it retains the possibility of moving the local optimum closer to the global one. Therefore, a franchise agreement is simply a more generalized form of the international park agreement required for the acquisition of all development rights in a region. A franchise agreement could be used to specify the precise range of activities allowed and disallowed in the region, and it would require the international community to provide the "global premium" that is necessary to make up the difference between what an entity would bid for the franchise and what the value of the land would be in sustainable use.

Market regulation agreements

The alternative to the direct purchase of development rights is the subsidization of diverse resource production, in order to alter the perceived benefits of conversion. The owner-state considering conversion will balance the comparative benefits from the land in its various uses. One means by which the decision-making process might be biased towards the naturally occurring slate of resources is the enhancement of the returns from these resources.

This approach obviously will not address all three of the potential routes to extinction (discussed in chapter 3), but it will address two of them. The third route—the mining of high value, slow-growth resources such as hardwood forests—must be addressed via the subsidies approach developed above. For all other resources threatened with extinction (those suffering from over-exploitation and under-cutting), a policy of *rent maximization* and *appropriation* is available.

Rent appropriation is a policy based on assuring that the owner-state receives the full value from its diverse resources. At present, this is the opposite of what is occurring with respect to a wide range of these resources. African countries were capturing about 5 percent of the value of their raw ivory exports during the height of the ivory trade (Barbier et al. 1990). Tropical bird harvesters around the world acquire between 1 percent and 5 percent of the wholesale value of the animals (Swanson 1992). This is true even with regard to many exports of tropical forest products (Repetto 1990). The owners of diverse resources under these circumstances are holding only the legal, and not the beneficial, rights of ownership. Combining the two sets of rights within the same entity will greatly enhance the perceived benefits from diverse resource management.

At present, international policies regarding diverse resource trade are the opposite of a rent appropriation system. International policy regarding poorly managed resources (for example, where the beneficial interest is separated from the legal resulting in over-exploitation) is to attempt to destroy the value of the resource (as discussed above with regard to CITES) or to bring the resource into domestication (as with the relocation of the livestock for the tropical bird industry to developed countries). Both approaches are geared equally to the destruction of incentives for investments in diverse resource habitats.

An international policy regarding diverse resource trade must instead be based upon a constructive approach by being used for the *maximization of the rental value* of the resource combined with the *targeted return* of that value to those states investing in their diverse resources. This approach provides compensation for those states already investing in the management of their diverse resources, and it provides incentives to those states not so investing.

For example, the CITES ban on the ivory trade can be seen as a misconceived policy. Although the continental populations of the elephant had declined by half, these populations had fallen precisely in those states that had not invested. In other states that had

invested heavily in the species, such as Zimbabwe (with per kilometer investments ten or twenty times as great as those in the non-investing states), the elephant population had vastly increased over the same period (by 100 percent in Zimbabwe). A blanket ban on the ivory trade provides the wrong incentive structure for the African states. The imposition of a ban proved that the states engaged in elephant over-exploitation were right; there was no future to be had from investments in this diverse resource.

The correct international incentive structure would do the opposite; it would provide *premiums* to the states investing in their diverse resources and *penalties* to those who do not. This could be achieved through a *sustainable wildlife trade exchange*. As with any exchange, it would be developed to discriminate between good (investing) and bad (non-investing) suppliers and allow only the former to sell on the exchange; then any consumer of the product would know that in purchasing from the exchange, the funding would flow to an owner-state that invests in the resource. In addition, to the extent that the consumer states agreed and enforced purchases solely from the exchange, the result would be greatly enhanced prices for the supplier states listed on the exchange. This *exchange-based price differential* would then constitute the premium for investing owner-states and the penalty for the non-investing.

The crucial element in this approach is the creation of a price differential for investing owner-states through market regulation in the consumer-states. Restrictions on the country from which consumers are allowed to make their purchases will always create a premium for the favored countries. A *global premium* for the sustainable producers of flows of diverse resources may be created through any scheme generally directed to this purpose.

The method of certifying suppliers in a wildlife trade exchange is the manner in which investments in diverse resources are assured. A host-state would only be "listed" on the exchange if it were able to demonstrate that its supplies to the exchange were derived from "sustainably managed habitats." It is the latter requirement that ensures that the proceeds received from the exchange are re-channelled to the habitat, because sustainable management requires substantial investments of resources (which is why it so seldom exists). It would only be those states that are willing to invest in this manner of development, and able to demonstrate their capabilities to do so, that would be listed on the exchange. In this way the exchange system generates the funding required for enhancing diverse resource benefits, and also generates the incentives for channelling these enhanced benefits back into the diverse resources.

Another example of such a scheme would be a *genetic (intellectual property) right regime*. This regime would also allot specific markets in consumer states to compensate for diverse resource investments, but the connection between the market and the investment would be less direct (as compared with stock investments that directly generate tangible flows).

The role of any form of intellectual property rights regime is to provide a basis for compensating investments in stocks that do not generate directly compensatory flows. Specifically, intellectual property regimes generally reward inappropriable investments in information with rights in discrete markets. A concrete example is the innovation of the optimal sized racquet head, developed from a program to determine the optimal trade-off between wind resistance (too large a head) and required accuracy (too small a head). The inventor of the oversized tennis racquet determined that a racquet of 117.5 square centimeters was optimal for tennis. In fact, this represented an investment in the creation of pure information that would not have been appropriable through the marketing of tennis racquets (because other sellers would immediately have entered the market with the same head). Therefore the intellectual property rights regime awarded this inventor with a protected market right in all racquet head sizes between 100 square centimeters (the original size of a tennis racquet head) and 135 square centimeters. This protected market then acted as compensation for the investment in the information created by this inventor.

It is equally possible to link protected markets to investments in diverse resource stocks, because these

stocks also feed into various industries in an indirect and usually inappropriable fashion. For example, many pharmaceutical innovations are developed from a starting point of knowledge derived from the biological activities of natural organisms. However, after the long process of product development and introduction, there is no compensation for the role played by the diverse resource in initiating the process.

A genetic resource right system could be constructed that would be analogous to an intellectual property rights system. This would require a royalty payment to the owner-state investing in the maintenance of diverse resources that are made available for prospecting by various industrial concerns. This royalty would be based on a protected market right for the return of some share of the revenues from the marketed product to the investors in the resource that led to the creation of the product.

In summary, the idea of consumer market agreements is to allocate these markets only to those owner-states investing in their diverse resources. The owner-states that choose to mine their diverse resources will otherwise drive down the prices, and rents, available to all states providing diverse resource flows. An agreement to restrict consumer markets to those owner-states that invest in their diverse resources creates a price differential: a price premium target to all sustainable producers and a price penalty target to all non-sustainable producers. Such a mechanism might be used in a wide variety of circumstances, where the stock-related investments are directly linked to the final product (for example, an ivory exchange), and where the stock-related investments are less directly linked to the final product (for example, a genetic resource right regime).

How are these considerations relevant to the Global Environment Facility? Since the GEF operates via projects, the concept of tradable development rights, whereby nation states capture what we have termed the global premium, does not seem appropriate for its remit. But over the long run the nature of the GEF is likely to change. One scenario is that it becomes as much a "broker" of some or many of the different ways in which international transfers take place. The GEF does deal directly in one mechanism for creating global premiums by changing the rate of return to conservation in the host country. As expertise is gathered in these ventures, the GEF might readily become an intermediary for private sector investments—a natural development of current co-financing agreements. For example, the GEF might monitor and "authenticate" investments by the private sector in country A to reduce CO_2 emissions in country B, an investment that would be justified for the industry concerned if (a) CO_2 quotas are imposed in the developed world; and (b) the costs of abatement are lower in the developing world. Extending this brokerage function still further, the GEF might ultimately involve itself in franchise and tradable development rights (TDR) agreements. This is a scenario for the future, not a prescription. But the kind of economic analysis in this section does suggest that biodiversity conservation will require more imaginative use of the limited resources available under the Rio conventions.

Domestic policies

Chapter 3 argued that land-based biodiversity erosion arises from under-investment and niche competition. Under-investment reflects the unappropriated global externality from biodiversity loss, which is the failure to capture global values, or the global premium. Chapter 2 showed that these premiums could be very large, as illustrated by the carbon storage values of tropical forests.

But chapter 3 also showed that even the *apparent* rates of return to land-use conversion are distorted by domestic policies. Correcting those policies thus becomes an integral part of the measures needed to reverse the decline in the world's stock of biodiversity. The general theme, then, of the policy measures emerging from the economic analysis of biodiversity loss is to:

- Establish a *domestically level* playing field between alternative land uses by removing market distortions in the form of subsidies and poorly defined property rights
- Capture the global premium to ensure that there is a *globally level* playing field
- Invest more in population control policies and in technologies to meet the needs of expanding populations currently met by land conversion.

To be sure, the final result will not be *total* conservation of biodiversity. If we knew all of the relevant information there might still be an "optimal" level of biodiversity loss. But it will be a marked change of emphasis compared to the current situation.

What kinds of domestic policy changes are required? We illustrate this briefly since extensive analysis already exists (Pearce and Warford 1993; Repetto 1986; Kosmo 1989; and Panayotou and Ashton 1992, among others). Essentially, removal or reduction of economic distortions would be beneficial to the economies of the country in question and simultaneously benefit the environment, and hence biodiversity in general.

In the developing and developed world alike, free markets are often not allowed to function. Governments intervene and control prices. In the European Community, agricultural prices are kept above their market equilibrium, with resulting over-production and damage to the environment through hedgerow removal and over-intensive agriculture. In the developing world the tendency is to keep prices down, below their market equilibrium. These interventions often cause environmental problems through the following negative effects:

- Governments use up substantial tax revenues and other income in subsidies for price control, even though government revenues are at a premium because of the need to use them to develop the economy.
- Subsidies encourage over-use of the resources that are subsidized. The effect of keeping prices down is to encourage wasteful use.
- Subsidies make the economic activity in question appear artificially attractive. This tends to attract more people into that industry or sector because profits, or "rents," are high. This is termed rent-seeking, and diverts resources away from more productive activities in the economy.

The impact on the environment can be illustrated in the context of the pricing of irrigation water and energy.

Irrigation water

In many countries the price charged for water that is used for irrigating crops is generally below the cost of supply, and often leads to a lack of incentives to conserve water (for example, charges are often set on the basis of irrigated acreage regardless of water quantity consumed). One of the effects of such low charges is over-watering, with the result that the irrigated land becomes waterlogged. Applications of irrigation water often exceed design levels by factors of three. In India, 10 million hectares of land have been lost to cultivation through waterlogging, and 25 million hectares are threatened by salinization. In Pakistan, some 12 million hectares of the Indus Basin canal system is waterlogged and 40 percent is saline. Worldwide, some 40 percent of the world's irrigation capacity is affected by salinization. Irrigation from river impoundments has resulted in other environmental effects. Large dams produce downstream pollution and upstream siltation as the land around the reservoir is deforested. Indigenous peoples are moved from their traditional homelands when the dammed area is flooded. Clearly, not all damage done by irrigation is due to low pricing, nor, by any means, can the environmental costs of large dams be attributed entirely to inefficient pricing. But there is an association between wrong pricing and environmental damage. By adopting prices that are too low, more irrigation water than is needed is demanded, exaggerating the requirement for major irrigation schemes such as dams, as well as for other schemes. Even if the scheme is justified, the amounts of water that are used are likely to be excessive because of the failure to price the resource closer to its true cost of supply.

Table 4.1 shows the actual revenues obtained from selected irrigation schemes as a percentage of operating and maintenance costs (O+M) and of total costs (capital conservatively estimated plus O+M). While some countries succeed in recovering most or all of the O+M costs, the highest recovery rate of total costs is only around 20 percent.

The under-pricing encourages a wasteful attitude so that systems are kept in a poor state of repair. Inefficient irrigation negatively affects agricultural output. Low charges lead to excess demand, giving a premium to those who can secure water rights, for example, by being the first in line to receive water. This is brought about because the system irrigates particular parcels of land first, leaving the poorer

farmer to secure whatever remains after wasteful prior uses. Moreover, water tends to be allocated according to acreage, not by crop requirements. This results in rent-seeking: the interest is in securing control of the allocation system. The high rents get capitalized in higher land values, making the incentive to compete for the allocation more intense. But the competition does not occur in the marketplace. It manifests itself as bribery, corruption, expenditures on lobbying, political contributions, and so on. The allocators of rights similarly expand their own bureaucracies and secure benefits for themselves. Rent-seeking obviously favors the already rich and powerful and discriminates against the poor and unorganized. And because it encourages wasteful use of resources, rent-seeking harms the environment, adding to the social costs of policy failures in the price-setting sphere.

Energy

Commercial energy forms such as coal, oil, gas and electricity are widely subsidized in developing countries. As with irrigation water, the effects of the subsidy are to encourage wasteful uses of energy, and therefore to add to air pollution and problems of waste disposal. The economic impacts of the subsidies tend to be more dramatic, since they are a drain on government revenues and divert valuable resources away from productive sectors; they also tend to reduce exports of indigenous energy, thereby adding to external debt, and encouraging energy-intensive industries at the expense of more efficient industries.

There are two measures of subsidy. The *financial* measure indicates the difference between prices charged and the costs of production. An *economic* measure indicates the difference between the value of the energy source in its most productive use (the "opportunity cost value") and its actual price. A convenient measure of the opportunity cost value, or "shadow price," is either (a) the price the fuel would fetch if it were exported, or the price that would have to be paid if it were imported (the "world" price), or (b) if the fuel is not tradable (as with most electricity, for example) the long-run marginal cost of supply. This long-run marginal cost of supply is the cost of providing an additional unit of supply in the long-term. The financial measure reflects the direct financial cost to the nation of subsidizing energy, but the economic measure is more appropriate as an indicator of the true cost of subsidies since it measures what the country could secure if it adopted a full shadow pricing approach.

Table 4.2 shows the size of the economic subsidy for selected oil-exporting countries. Here the subsidies have an additional distortion in that they divert potentially exportable energy to the home market, thus adding to balance-of-payments difficulties and hence to international indebtedness. The scale of the distortion can be gauged by looking at the subsidies as a percentage of energy exports and as a percentage of all exports. In Egypt, for example, the subsidies are equal to 88 percent of all exports and twice the value of oil exports.

Governments are very often themselves the cause of environmental degradation. While we are all used to the idea that governments should put things right, we are less familiar with the idea that certain government policies, even those that ostensibly have nothing to do with the environment, can, and often do, damage the environment. This is "government failure." Clearly, since markets fail too, the issue for policy is to find the proper balance between the role of markets and government intervention.

Table 4.1 Cost recovery in irrigation schemes
(percent)

Country	*Actual revenues O+M Costs*	*Actual revenues Capital + O+M costs*
Indonesia	78	14
Korea	91	18
Nepal	57	7
Philippines	120	22
Thailand	28	5
Bangladesh	18	neg

Notes: neg = negligible. Capital costs are "moderate" estimates only.
Source: Repetto (1986)

Table 4.2 Economic subsidies to energy use in selected countries

	Size of subsidy $m	*As % value of all exports*	*As % value of energy exports*
Bolivia	224	29	68
China	5400	20	82
Egypt	4000	88	200
Ecuador	370	12	19
Indonesia	600	5	7
Mexico	5000	23	33
Nigeria	5000	21	23
Peru	301	15	73
Tunisia	70	4	10
Venezuela	1900	14	15

Source: Kosmo (1989)

Community involvement

Whatever the national or global value of biodiversity conservation, its size will be generally irrelevant if those values are not appropriable by the individuals making land-use decisions, whether they be loggers or squatters, permanent agriculturists or ranchers. Wells (1992) notes that the benefits of biodiversity protection through national parks tend to be lowest at the local level and highest at the national and global levels (see chapter 2 as well). But when analyzing costs, they are highest at the local level and lowest at the national and international levels. As such, the *net benefits* of conservation are lowest for the local community and highest for the national and global community. Indeed, at the local level, net benefits may be negative, indicating that there is no local incentive to undertake land conservation.

This suggests that not only must the local community be involved in conservation efforts (now a standard policy prescription) but that they should also be able to appropriate a fair share of the wider values of conservation. But even where these two conditions of *involvement* and *net local gain* are met, it cannot be assumed that conservation will be undertaken. Wells and Brandon (1992) note some additional requirements, especially the need to account for the state of the local economy. For example, merely providing an alternative, sustainable land use such as agroforestry may not result in reduced deforestation if the local economy is characterized by surplus labor. The effect may simply be that the new land use is absorbed by the surplus labor, leading to earlier levels of land use (as with some coca "replacing" projects; see Southgate and Clark (1992)). Demonstrating and marketing the value of a sustainable product could even backfire when property rights are weak, as with the overexploitation of the fruit aquaje round Iquitos in Peru. Previously sustainable picking of this fruit gave way, through market development, to rent-seeking and felling of the trees containing the fruit.

How are such pitfalls in well-meaning investments to be avoided? Careful design of the context of the investment is critical. The likely reactions of the local community to such investments need to be carefully gauged, including the very real potential for rejection of the project as an invasion of existing rights. Local communities must be able to identify increased rents from the conservation activity compared to the existing returns from the mining of renewable resources. As Southgate and Clark (1992) point out, rents for many agricultural colonists at the forest edge are zero, since the colonists cannot influence price (they are "price takers"). What they receive for, say, timber is an amount approximating what they can earn by applying their labor elsewhere (the "opportunity cost" of labor), and this will be considerably less than the market price of the timber. This accounts for the divergence between actual land prices in such areas and the land price that would result if owners could capture all of the market value. All this suggests that investments in conservation must ensure that the rents from conservation accrue, in significant part, to the local community that will be involved in implementing the conservation activities. If Wells (1992) is right, conservation projects may sometimes yield *negative* rents for local people, making the project even less attractive than the meagre zero rent activity they usually engage in.

But there is an additional policy measure, namely, investing in the activity that gives rise to the biodi-

versity loss. While this may seem contradictory, what is involved is raising the productivity of lands *outside* the areas where biodiversity is to be conserved through sustainable use activity or outright protection. As chapter 3 showed, one of the fundamental forces at work in explaining land conversion is population growth and density. But the demand for "new" land could be reduced by raising the productivity of existing land through measures such as agricultural investment, extension and irrigation. Instead of focusing solely on investment in the protected area, the focus should also be on raising agricultural productivity to reduce the *motivation* for land conversion.

This approach also avoids or mitigates the difficult problem of choosing between biodiversity investment projects—a basic concern for the GEF—for it suggests that more will be achieved by agricultural development, and fuelwood substitution technologies, than by protected areas. If the GEF is to succeed in biodiversity conservation, conventional development assistance needs to be strengthened with respect to the agricultural sector.

5 Summary and Conclusions

The economic value of biodiversity

Why conserve biodiversity? There are three potential answers:

(a) The constituent parts of biological diversity have some intrinsic right to exist, a value in themselves, independent of human valuation.
(b) The erosion of biodiversity threatens the well-being of the human race, regardless of any intrinsic concept of value.
(c) Humans wish to conserve biodiversity, a wish they express through lobbying, a willingness to pay, and so on. This valuation may be independent of any belief about intrinsic value or any risk assessment of biodiversity erosion.

In reality, human valuations of biodiversity are likely to reflect all three motivations. Distinguishing between them can be complex, or even impossible. This paper is primarily concerned with motives (b) and (c), motives for what we term *economic valuation.*

The concept of economic value

Total economic value (TEV) can be broken down into use and non-use values, the latter being measured by a willingness to pay for conservation unrelated to any use, now or later, of the resource. Use values comprise direct uses (such as harvesting and tourism), indirect use values (for example, habitats as carbon stores and watershed protection assets), and option values (an insurance premium to ensure future use).

Of fundamental importance to GEF is the broad division of TEV into "domestic" and "global" values. The former accrue to individuals within the host-state, the latter to the rest of the world. Thus, at any stage there is a potential eight-fold categorization of economic value, as in Table 5.1. GEF is

Table 5.1 Economic value classification

	Use value			*Non-use value*
	Direct	*Indirect*	*Option*	*Existence*
Domestic	Harvesting	Watershed protection		WTP for conservation in own country, unrelated to use
Global	Ecotourism	Carbon store		WTP for conservation in other countries as revealed in debt-for-nature swaps, etc.

concerned with (a) determining global value; and (b) seeking means whereby host countries can benefit from global value.

Preliminary investigations suggest that global value, particularly global indirect use value, and perhaps existence value, are large relative to the domestic returns from land conversion. Thus, carbon storage values in tropical forests may be as high as $500 to $2,000 per hectare.

Why are we losing biodiversity?

Land-based biodiversity is being lost due to two fundamental forces:

(a) Under-investment; and
(b) Niche competition.

These forces are different for land and water. Many water resources are open access resources, not being owned by anyone. The traditional "tragedy of the commons" argument does much to explain the loss of international water-based biodiversity.

But land-based resources share the open access features of water only because nation states choose not to invest in those resources. All land-based resources have state "owners." International waters do not always have owners. To explain land-based biodiversity loss, then, we need to explain under-investment in biodiversity.

Under-investment

This arises because the "rate of return" to conservation and sustainable use is less than the rate of return to land conversion. This is so for three reasons:

(a) Even where there are no deliberate attempts to distort the functioning of markets, and where there is no global value, some land conversion beyond the socially desirable amount will occur. This is due to *market failure*—the failure of markets to account for the side effects of the land conversion, such as downstream sedimentation and loss of biodiversity. The *true rate of return* to land conversion is less than the perceived rate of return (in economics jargon, *externalities* have been ignored).
(b) Land conversion is often subsidized through direct grants for land clearance, subsidies for credit, agricultural inputs, and land purchase, or through the maintenance of exaggerated prices for agricultural output. Land tenure and resource rights are often ill-defined for the sustainable uses of land, and may be made secure by the conversion process. The excess amount of land conversion that occurs because of these factors reflects *government failure*—inefficient interventions in the marketplace by governments.
(c) Some of the external benefits of sustainable land use are *global* and are not captured by the "owners" of land, whether this is the immediate landowner or the government. Since these global benefits (such as carbon storage) are not under the control of the nation-state, they are not appropriated, and hence do not appear as a domestic benefit to land conservation.

These domestic market, global market, and government failures help to explain why the rate of return to sustainable uses of land is below the rate of return to land conversion. There is no *level playing field* between conservation and conversion.

Niche competition

The sheer expansion of human numbers has and will place pressure on the available unconverted land. Combined with under-investment in conserved habitats, and hence in biodiversity, niche competition produces a fairly relentless demand for land conversion. Statistical analysis tends to support the significant role of population density as a factor explaining conversion and, to a lesser extent, population growth. But other factors are also at work, including national indebtedness to some extent, and certainly income growth.

Towards a biodiversity policy

Once the causal factors giving rise to biodiversity loss are put together in this overall picture, the directions for policy to slow the rate of biodiversity loss become clear, though complex to implement:

(a) Continued efforts to slow population growth need to be emphasized so as to reduce the competition for available niche space.

(b) A major focus needs to be on measures which change the rates of return to land use, upwards in the case of biodiversity and downwards in the case of conversion. There are many measures which can be employed:

- Continued pressure on domestic governments to reduce and remove economic distortions such as conversion and input subsidies, and guaranteed output prices. While some subsidies reflect deliberate policy to meet the needs of the poor, many are not targeted in this way and accrue to the wealthier classes of society.
- Land registration, titling and resource rights for those practising sustainable land use.
- Mechanisms to capture the global benefits of sustainable land uses. Various candidates here are:

  (i) Global Environment Facility: the return to conservation investment projects is inflated to reflect the global benefits they generate.
  (ii) Franchising agreements: land use is restricted in return for payments from some internationally agreed fund. Various forms of franchising agreements are possible.
  (iii) Debt-for-nature swaps.
  (iv) Tradable Development Rights (TDRs): domestic or international purchasers might buy development rights to zoned land in a given country. In exchange for payment, land users in conservation zones forego the right to develop the land in a manner inimical to biodiversity conservation. The price of the TDRs thus reflect the foregone value of converting the land—the "opportunity cost" of conservation. As discussed in chapter 4, purchasers may be governments, but could more interestingly be environmental organizations and the private sector.

A schematic summary

Figure 5.1 summarizes the essential features of this report, and shows the links between valuation, causation of biodiversity loss, and remedial measures.

The role of the Global Environment Facility

Figure 5.1 shows the context in which the GEF operates. Clearly, the GEF has a very limited role to play with respect to niche competition due to population growth. Policies to control population growth are of the utmost importance but lie within the remit of existing national government policies assisted by international agencies. The focus of GEF activity therefore has to be on under-investment in land uses to conserve biodiversity, and that focus has to be within the context of nation-state priorities and conservation strategies.

All nations-states attenuate the use of land in one way or another. As far as its biodiversity activities are concerned, the GEF has so far operated mainly by raising the rate of return to protected areas. It conserves *existing protected areas* by reinforcing existing zoning policies, which in turn attenuate the development uses of land. For a totally protected area, the development uses are totally or near-totally attenuated. In order partly to compensate for foregone development values, the rate of return to protection is raised through global transfers of resources to pay for the costs of protection. The payment is not a subsidy, but a transfer in return for which the rest of the world secures a benefit in the form of conserved globally important biodiversity. In this mode, GEF is, in terms of Figure 5.1, in fact executing an *international franchise agreement.*

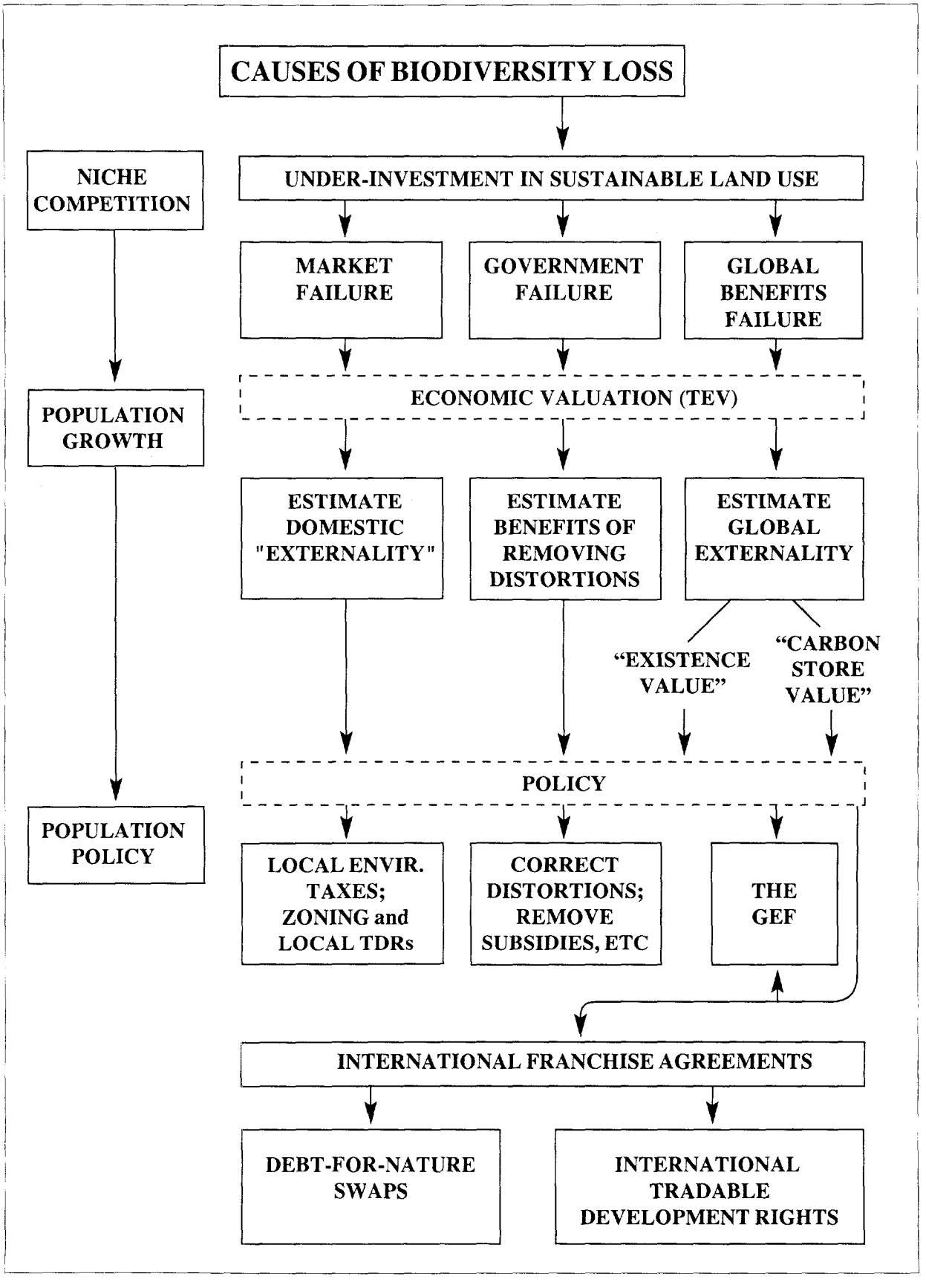
One issue that arises, then, is the extent to which GEF should extend its activities to *preventing future land conversions* that would not be justified through a full global cost-benefit appraisal. As an example, consider the forces giving rise to future deforestation, or future drainage of wetlands, or the ranching of wildlands. The rationale for this kind of intervention is two-fold:

(a) GEF is currently the only international fund that seeks to capture global benefit;[4] and
(b) It may be a very cost-effective way to conserve biodiversity.

Indeed, there is evidence to suggest that, in South America at least, significantly less land conversion would occur if investments were targeted not at protected areas, but at the areas *outside protected areas.* Southgate (1991) has shown that *raising agricultural productivity* could significantly reduce the drive by

[4] Debt-for-nature swaps do this but are piecemeal and financially very small compared to the GEF.

Figure 5.1 Schematic summary of factors affecting global biological diversity



agricultural colonizers to expand into forested areas. Such a policy fits the under-investment hypothesis since it addresses the fundamental disparity between the rate of return to colonizers from land conversion and existing land intensification. But such projects are more typically the province of conventional development aid. They may also generate a "magnet" effect by attracting inward migration which would then threaten the forest areas.

How far such a suggestion would change the remit of the GEF is not clear. Raising agricultural productivity in buffer zones would fit into a general conservation strategy, but more generally raising agricultural productivity on degraded lands as a focus of a project is less likely to figure in the GEF portfolio. In the Pilot Phase, few projects have appeared to emphasize the preventive approach. In GEF II, the fully operational phase, with its greater emphasis on cost-effectiveness, it is arguable that this imbalance should change. At the very least, the relative cost-effectiveness of the existing-areas focus versus the projected-land-conversion focus needs to be explored.

The role of GEF in seeking to change economic distortions also raises difficult questions. By and large, "conditionality" clauses in conventional aid programs cover the more obvious examples, and their controversial nature must be acknowledged. There are at least two reasons why the GEF need not consider conditionality as part of its project evaluation procedure:

(a) GEF projects tend to complement existing aid projects, so that they have limited "stand alone" features; and

(b) GEF projects are designed to capture global benefits, so the idea of seeking conditions for resource transfers to benefit the donors seems contradictory.

Finally, there is the important question of other roles for the GEF. It seems clear that the most effective way forward for the GEF is to evolve away from being only a project focused agency. GEF funding is unlikely to be at levels where major contributions to biodiversity conservation can be made. But the GEF can leverage other funds, and it can act as a broker for other forms of transfer such as international tradable development rights and international franchising agreements.

As noted in chapter 4, there are several reasons why purely private sector transfers could take place to conserve biodiversity. In these transfers, the GEF could act as a monitoring, brokerage and authentication agency, thereby widening its role and becoming more central and more effective in international efforts to conserve biodiversity. But any new role, for example, in relation to franchise agreements, could involve conditionality. The payment of the "global premium," say annually, by the franchisee, would be conditional on evidence that the agreed attenuation of land use had been honored. The conditionality would thus be between the recipient of the premium, the nation-state and the agent to whom the franchise is let. The GEF could act as the authenticating agent to ensure that the agreement is being honored (on both sides).

Appendix I

A Survey of the Results of Econometric Studies on Deforestation

Study	Type of Analysis	Dependent Variable	Independent Variables					
			Population	Population Density	Income	Agricultural Productivity	External Indebtedness	Other Significant Variables
1. Shafik, 1992[b]	the causes of deforestation for a sample of 66 countries	annual rate of def. 1962–86						investment rate— positive electricity tariff— negative trade shares in GDP— negative political rights— positive civil rights— positive
2. Burgess, 1992	def. in 53 tropical countries using cross-sectional analysis.	five-year change, 1980–85, in closed forest area.		negative 0.01	(real GNP per capita in 1980) positive 0.05			roundwood production per capita, 1980— negative, 0.05. the log of closed forest area as a percentage of total forest area in 1980— positive, 0.1
3. Burgess, 1991[c]	def. in 44 of the major tropical forest countries using cross-sectional data.	model 1. level of def.	(pop. growth) negative 0.05		(GDP per capita) positive 0.05		(debt-service ratio as a % of exports) positive 0.05	total roundwood production— positive, 0.05
		model 2. level of def.	negative 0.01				positive 0.10	food production per capita— positive, 0.10 total roundwood production— positive, 0.05.
4. Southgate, 1991[d]	causes of agricultural colonization in 23 latin American countries.	growth in the area used to produce crops and livestock.	pop. growth positive 0.01			negative 0.01		agricultural export growth— positive, 0.05
5. Kahn & McDonald, 1990[e]	develop model to show economic mechanisms by which debt may lead to def.	def. area (1000 ha)	negative 0.017 (significant alpha level). pop. used as a proxy for the labour force				positive 0.01	forested land area— positive, 0.01 annual change in public external debt— positive, 0.05

			Independent Variables					
Study	Type of Analysis	Dependent Variable	Population	Population Density	Income	Agricultural Productivity	External Indebtedness	Other Significant Variables
6. Capistrano & Kiker, 1990[f]	global economic influences on tropical closed broad-leaved forest depletion, 1967–85.	model 1: def. (the area of closed broad-leaved forest depleted by commercial logging).			(per capita) positive 0.01 P.(period) 2		(debt-service ratio)	log export value— positive, 0.01, P.1. real devaluation rate— positive, 0.05, P.3&4. cereal self-sufficiency ratio— positive, 0.01, P. 2.
		model 2.			positive 0.05 P.3		negative 0.05 P. 2	log export value— positive, 0.01, P.1. agri. export price index — positive, 0.05, P.2. real devaluation rate — positive, 0.01, P.3 / 0.05 P.4. cereal self-sufficiency ratio— positive, 0.01, P.2 / 0.05 P.3. arable land per agricultural capital—positive, 0.01, P.4.
		model 3.	positive 0.05 P.2		positive 0.01 P.2		negative 0.05 P.2	log export value— positive, 0.01, P. 1. agricultural export price index— positive, 0.05, P.2. real devaluation rate— positive, 0.05, P. 3. cereal self-sufficiency ratio— positive, 0.01, P.2 / 0.05, P.3. arable land per agricultural capita—positive, 0.01, P. 4.
7. Rudel, 1989[g]	decline in closed tropical forests for 36 countries across Africa, Asia and Latin America.	average annual decline in hectares of a country's tropical forests during the period 1976–80	pop. growth positive, 0.001 (rural pop. growth, positive, 0.01)		GDP per capita. positive 0.01			forest land area— positive, 0.001.
8. Paulo, Mery & Salmi, 1987	cross-sectional test of factors influencing deforestation in 72 countries.	absolute forest cover in 1980.		negative. 0.01				food production per capita— negative, 0.01 share of forest fallow— positive, 0.1 agri. area coverage— negative, 0.1

Study	Type of Analysis	Dependent Variable	Independent Variables					
			Population	Population Density	Income	Agricultural Productivity	External Indebtedness	Other Significant Variables
9. Allen & Barnes. 1985[h]	def. between 1968–78 in 39 countries in Africa, Latin America and Asia.	model 1: annual change in forest areas.	pop. growth negative 0.10					
		model 2: the decade change in forest area, 1968–78.						logarithm of % of forest cover 1986— positive, 0.10. the % area under plantation crops in 1968— negative, 0.05 per capita wood fuels consumption and wood exports in 1968— negative, 0.05
10. Lugo, Schmidt & Brown, 1981	def. in all greater Caribbean countries.	% forest cover		negative 0.001				energy use per unit area— positive, 0.001
11. Reis & Guzman, 1992[i]	Brazilian Amazon deforestation and its contribution to CO_2 emissions.	def. density.				positive 0.001		cattle herd— positive, 0.05 logging— negative, 0.01.
12. Katila, 1992[j]	def. in Thailand	relative forest cover by country		negative 0.01		negative 0.05		wholesale price of construction timber— negative, 0.01.
13. Constantino & Ingram, 1991[k]	def. rates in Indonesia.	relative forest cover.		negative 0.01	GDP per capita positive 0.05	positive 0.01 (rice production used as a proxy)		time— negative, 0.01

Study	Type of Analysis	Dependent Variable	Independent Variables					
			Population	Population Density	Income	Agricultural Productivity	External Indebtedness	Other Significant Variables
14. Kummer & Sham, 1991[l]	def. in post-war Philippines.	1. cross-sectional analysis for the years 1957, 1970 and 1980. absolute amount of forest cover per province in hectares.		negative 0.05 1970 and 1980.				road density— negative, 0.05, 1957, 1970 & 1980 kilometers of road— positive, 0.05 1980.
		2. panel analysis. absolute loss of forest cover, 1970–80 per province in hectares						forest area— positive, 0.05 distance from Manila— positive, 0.05 logging in 1970— positive, 0.05.
15. Panayotou & Sungsuwan 1989[m]	def. in N-E Thailand.	forest cover		negative 0.01	positive 0.01 (provincial income)			wood prices— negative, 0.01 distance from Bangkok— positive, 0.01 rural roads— negative, 0.10 rice yields— positive, 0.10 model 2: price of kerosene— negative, 0.01.
16. Southgate, Sierra & Brown, 1989[n]	def. in 20 cantons in Eastern Ecuador in early 1980s.	def.	(agri. population) positive 0.05					tenure security— negative, 0.06.

Notes

[a] Presentation: studies explaining deforestation globally are listed first (entries 1-9), followed by country and regional studies (entries 10-15). The sign, i.e., positive or negative, and significance level, where available, is indicated for each significant independent variable.

[b] Panel regressions are used to test three models—log linear, quadratic and cubic. Shafik estimates both the annual and total measures of deforestation (only annual deforestation results are presented). He concludes that per capita income has virtually no explanatory power, statistically speaking, in both cases regardless of the functional form. (Per capita income is defined as real per capita gross domestic production in terms of purchasing power parity.)

[c] 1.Model 2 includes two dummy variables to capture possible regional differences between African, Asian and Latin American countries.
2. Food production is used as a proxy for food demand.

[d] Explores the possibility that deforestation in Latin America is symptomatic of agricultural underdevelopment. Southgate argues that since property arrangements oblige agricultural colonists to pay scant attention to the value of tree covered land, the option of using the ratio of cleared area to remaining forest makes little sense. The growth in the area used to produce crops and livestock is therefore chosen as the appropriate dependent variable for causal analysis of frontier expansion.

[e] 1. Results shown are for 1981-85 data.
2. The negative coefficient for population, used as a proxy for labour input, stems from the authors' definition for GNP. A greater labour force leads to a higher GNP, thus reducing the need to deforest to meet current consumption needs.
3. The regression results show a strong relationship between debt and deforestation. To eliminate the possibility that this might be a function of country size and not a real behavioural relationship, the regression is repeated with deforestation; debt and forest land area are defined in per capita terms (results not shown in the table). In the scaled regression, the public debt variable is the most significant behavioural variable.

[f] 1. Three linear models are estimated:
- Model 1: OLS
- Model 2: a fixed effect model with the same explanatory variables as model 1, but with an intercept term allowed to vary by regions, income class and international credit standing
- Model 3: intercept is constrained to be equal for all subgroups.

2. The period 1967-85 is divided into four subperiods:
- Period 1: 1967-71—the waning years of the system of fixed exchange
- Period 2: 1972-75—started with grain shortages, saw oil price increases and credit expansion, and ended with recession
- Period 3: 1976-80—booming commodity prices led to slow economic recovery, conditional lending and the second round of oil price increases
- Period 4: 1981-85—deep recession and painful adjustments as developing countries staggered under the burden of debt.

3. Period 1: tropical wood was the most significant variable, explaining more than 85 percent of forest depletion.

Period 2: cereal self-sufficiency ratio and the debt service ratio were the most significant variables.

Period 3: real devaluation rates had the strongest statistical relationship to forest depletion.

Period 4: expansion of arable land had the strongest influence on forest.

Results suggest that population has had a less direct impact on deforestation than macroeconomic variables.

[g] The results shown are weighted by the size of a country's closed forest area. This procedure makes equal units of forest area, belonging in varying proportions to different nations, the unit of analysis. GNP explains a substantial amount of the variation in the weighted analysis, but fails to show much variation in the unweighted analysis (results not shown). Rudel argues that the results demonstrate the importance of capital availability, GDP, on the deforestation of large forests through the funding of capital intensive projects. In countries with small scattered rainforests, encroachment by growing rural population is more relevant to the deforestation process.

[h] 1. In model 1 the coefficient for change in arable land is *not* significant. However, the authors observe that the bivariate correlation coefficients show that population growth is related to agricultural expansion, which is in turn related to deforestation. This relation does not show up in the multivariate analysis since controlling for population suppresses the negative correlation between arable land and forest loss; they therefore conclude that both population growth and change in arable land are associated with deforestation.

2. Model 2 is specified in order to capture the possible delayed impact of harvesting forests for fuelwood and wood exports on the rate of deforestation. While the coefficient per capita wood is not significant in the first model, it is in the second, suggesting that deforestation is significantly related to population growth and agricultural expansion in the short term, and with wood use (fuelwood and wood exports) in the long term.

[i] Population was found to be insignificant.

[j] Study concludes that population density is the most important cause of deforestation in Thailand.

[k] The time trend is used to capture the effects not accounted for, more specifically the cumulative roads built.

[l] The panel analysis does not support the contention that population growth is one of the leading causes of deforestation. Kummer argues that forest cover cannot be used as a dependent variable when analyzing ongoing deforestation because it cannot capture the dynamic nature of tropical forest removal. The cross-sectional analysis is therefore not concerned with deforestation per se, but with the relative absolute forest cover at one point in time with the hypothesized independent variables. The panel analysis is, however, directly concerned with deforestation since the dependent variable is the absolute change in forest cover. Results for the Philippines are said to support this by the fact that the cross-sectional and panel analyses yield such different results, with none of the independent variables in the panel analysis appearing in the cross-sectional equation.

The panel analysis cannot support the contention that population growth is one of the leading causes of deforestation.

[m] Two models are estimated to account for the high multi-collinearity between kerosene price, crop price and the price of wood. Model 1 includes the price of wood and crops, and model 2, the price of kerosene. The difference in the explanatory power of the models was found to be minimal. While the results for model 1 are presented in the table, the significance of the price of kerosene coefficient is noted. In model 1, population density emerges as the single most important cause of deforestation, followed by the price of logs and the distance from Bangkok.

[n] Under the hypothesis that settlement in tree covered hinterland is stimulated by the prospect of capturing agricultural rent, the authors first examine the relationship between rural population pressure (agricultural population) and the factors affecting agricultural rent, the scale of the urban population as a proxy for the local demand of agricultural commodities, soil quality, and road accessibility. Deforestation is then regressed against agricultural population and an index of relative tenure security among cantons.

Resource Franchise Agreements

Chapter 4 notes various forms of international resource franchise agreements. In each case, the general principle is that land use is reduced in a given zoned area in return for the payment of a premium. If all land uses other than protective ones are forbidden, the premium is equal to a rental on the land, and the donor effectively pays the rent on the land. In turn, this rent would be approximately equal to the rental that the land would command in some developmental use—technically the "highest" sustainable use value. If only some land uses are forbidden, then the premium will tend to converge towards the differential returns that land could have earned in the absence of reduced use.

Several franchise-type agreements have been discussed in the literature (Sedjo 1988, 1991; Panayotou 1992; Katzman and Cale 1990). The importance of international trading in such land-use rights ("development rights") is that it offers a means of capturing the global premium, the willingness to pay of the rest of the world for a nation's conservation. If all development is restricted, the *minimum supply price* should approximate the development value of the land since this is what is surrendered with the franchise agreement. If only some uses are restricted, the minimum supply price should be the difference between the overall development value and the returns obtained by operating the restricted uses. The *demand price* will be determined by the global willingness to pay for the global benefits. Chapter 2 suggested that for carbon, this demand price might be several times the total development value. This picture will, however, be influenced by discount rates. The global premium will be a regular payment, say annually, since it has to be conditional on performance. But the developmental land use value could be based on, say, clear felling the site in a single year. It is therefore the resource owner's discount rate that will be relevant when making this comparison between a stream of annual premiums and the development value of the land. The issue is complicated by the potential for successive uses of the land. Consider the "nutrient mining" sequence in a tropical forest (Schneider 1992). If markets work well, a logger seeking a few trees on a given hectare should pay for the land a price reflecting not just the rental on the logs but also the value of the successive uses of the land for crops and ranching, since the logger can "on-sell" the land to the agriculturist, and so on. If markets do not work well, the premium may be difficult to assess.

Who would buy such development rights? Panayotou (1992) indicates that the international markets would be local and international environmental organizations, governments, corporations and the scientific community (who would effectively buy the information value of the site). The motives for such purchases would vary:

(a) Environmental groups would be expressing their non-use ("existence") value for the sites;
(b) Governments might be expressing some existence value on behalf of their populations, but would certainly be the likely agents for expressing the global indirect use values such as carbon storage in forests; and
(c) Corporations might be motivated in several ways:

- Pursuing the tropical forest example, they might purchase conservation rights to forests in order to secure offsets to increased CO_2 emissions elsewhere, the offset being required because of some national CO_2 control target under the Climate Convention ("joint implementation").[5]
- They might wish to buy "exotic capital" to further a green image domestically or internationally.
- Some might wish to buy use rights to, say, pharmaceutical material.
- As Panayotou (1992) notes, they might also wish to speculate on the growth in the value of the tradable development rights (TDR) as the "demand for conservation" grows, that is, simply hold TDR for their asset value. The "carbon credit" value of a forest should also grow if carbon taxes in the developed world rise over time.

The incentives to sell the rights would be straightforward. It would pay the owner-state to sell any development right for a price higher than the foregone development value.

[5] There is a complication here. An offset could involve a *growing* forest to fix CO_2 released in the original country. Purchase of *existing* forests in carbon equilibrium begs the question of what would have happened if the development rights had not been purchased. Unless there is some certainty that the area would have been deforested there is no effective offset. One approach to this problem would be for an agency, such as the GEF, to determine the likelihood of "development." The potential for gains from threats should not, however, be overlooked.

Incremental Costs

This appendix outlines the basic elements of the measurement of incremental cost. It is important to understand that, in practice, estimating incremental cost is complex and must be adapted to the context of the institution and country in question.

We distinguish two contexts: a simple one in which the country in question has only one choice of technology, and a more complex one in which there is a choice of technologies. "Technology" here needs to be interpreted broadly. In the context of greenhouse gas control, it can refer to an energy source or to a carbon sink. In the context of the ozone layer, it refers to substitutes for chlorofluorocarbons (CFCs). In the international waters context, it may refer to different options for controlling waste; and in the biodiversity context it will refer to different ways of meeting a given protection objective.

Notation

In what follows we use C for cost; B for benefit; d for domestic or national; and g for global, where global means the rest of the world. IC is incremental cost, and δ means "difference in."

Simple case: single technology

In the simple case there are two possibilities:

(a) Bd > Cd

In this case, the domestic benefits exceed domestic costs—*prima facie*, therefore, the GEF would not be involved in financing the project. The country secures net gains by investing in the project itself or through conventional development aid sources. However, if there are major global benefits associated with this investment, it might qualify for GEF intervention as a Type I project.

(b) Cd > Bd

In this case, the country in question will not invest in the project since it secures net losses to the country. But if there are significant global benefits, GEF may wish to intervene. The requirement for its intervention is then:

$$Bg > (Cd - Bd) \quad ... [1]$$

This means the GEF may intervene to finance the project provided the global benefits exceed the *net cost to the nation if the nation had funded the project.* The amount Cd - Bd is the incremental cost.

What flows of funds are associated with this case?

(i) The country pays some of the cost since it gets a benefit Bd, but it does not pay all of the cost (otherwise it would have no interest in the project). The limit to the country's contribution is given by Cd *less* GEF's contribution which is Cd - Bd. So the country's contribution is less than Cd - (Cd - Bd) = Bd.
(ii) GEF pays, in the limit, Cd - Bd, which is the incremental cost.

This context is likely to define many of the biodiversity projects for GEF. Essentially, they will be projects where the country finds that the benefits to itself from conservation are not sufficient to justify conservation. Hence it will not proceed without GEF intervention. GEF must be satisfied that:

(a) Domestic benefits are not greater than domestic costs; and
(b) Global benefits exceed the incremental cost.

Multiple technologies

Cases where a country has several choices of technology may be less relevant to biodiversity, but the analysis of incremental cost is not complete without an assessment of this issue. Economic rationality from the country's standpoint dictates that it will choose the least-cost technology. But this may not be the most beneficial technology in terms of the global environment. For example, in the global warming context, the country may be able to burn coal or import gas. The coal is cheaper than the gas but emits higher amounts of CO_2. When should GEF intervene?

Let Cd_0 be the cost of the technology 0, and let this be the "cheap" technology. Let Cd_1 be the cost of the more expensive but more globally beneficial technology. Now the condition for GEF intervention is:

$$Bg > (Cd_1 - Bd) \quad ...[2]$$

This means that the global benefits must now exceed the net costs to the nation of adopting the *more expensive* technology.

Let $Cd_1 - Cd_0 = \delta C$, which is the difference between the costs of the two technologies. Then the requirement for intervention can be expressed as:

$$Bg > (Cd_0 + \delta C - Bd) \quad ...[3]$$

or

$$Bg > ([Cd_0 - Bd] + \delta C \quad ...[4]$$

This is the same requirement as for the simple technology case, but the term δC is added.

The whole expression on the right hand side is the incremental cost, IC.

If Cd_0 exceeds Bd the country will not proceed anyway, and since $Cd_1 > Cd_0$, it will not be interested in the globally cleaner technology either. So the only context of interest is the one where $Bd > Cd_0$, but $Bd < Cd_1$. That is, in its own interests, the country will proceed with the less globally beneficial technology and will not choose the more beneficial technology. But this means that $Bd > Cd_0$, in which case the first term on the right hand side of [4] is *negative.* The implication is that GEF should not seek to fund the complete difference in the costs of the two technologies (δC) but that difference *less* the net benefits the country would have got from proceeding with the less clean technology. The intuition here is that the GEF should not be paying anyway for the net benefits the country would have got.

What now are the resource flows?

(a) The total cost of the project is Cd_1 and the GEF would pay, in the limit, the amount this cost *less* an estimate of the benefits accruing to the nation; and

(b) The country would pay up to $Cd_1 - (Cd_1 - Bd) = Bd$, the benefits that it would obtain.

The results are thus similar in form to the simple case.

Of course, the preceding analysis assumes that benefits and costs are measured in the same units (such as money) and this will not always be possible. In some contexts, especially biodiversity and international waters, monetary assessment of benefits will be very limited. Hence a significant judgmental element will enter into the assessment of incremental cost since, as shown above, it must always involve an assessment of the domestic benefits.

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