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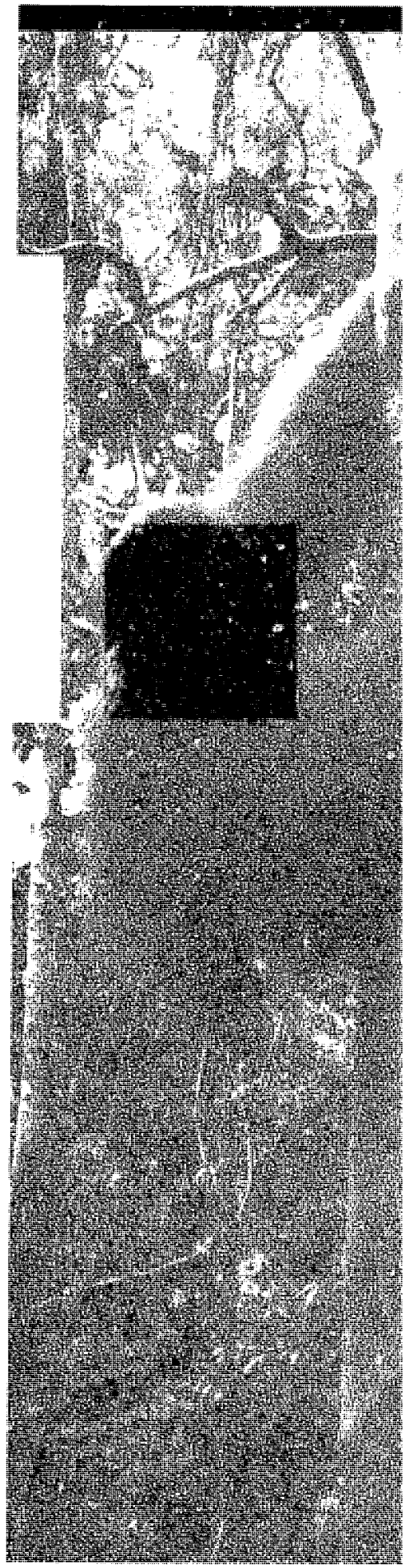
REPORT



THE CENTER FOR TROPICAL CONSERVATION

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BIODIVERSITY OF TROPICAL RAIN FORESTS: ECOLOGY AND ECONOMICS OF AN ELUSIVE RESOURCE

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SUMMARY

The study of the ecology and economics of tropical rain forest biodiversity leads to two major conclusions. First, ecologically and economically sustainable development alone will not stop the loss of tropical rain forest biodiversity caused by fragmentation, exploitation, and human activities. Therefore, strict biodiversity-preserving action is required. Second, while the costs of biodiversity preservation accrue mainly at the local level, the benefits accrue mainly at the regional, national, and especially global level. Therefore, special mechanisms must be devised to fund biodiversity preservation. These conclusions suggest the following priorities for applied research and action: study of forms of forest exploitation compatible with strict biodiversity protection; study of the magnitude and incidence of the benefits and costs of biodiversity protection; and, a search for ways of implementing integrated land-use complexes and other solutions.

Tropical Deforestation and the Global Biodiversity Crisis

Tropical rain forests and the biodiversity they contain are currently besieged by human-induced deforestation. Loss of habitat is compounded by the disturbance of remaining forest fragments and overexploitation through hunting and extraction. Populations of rain forest species tend to go extinct rapidly in isolated forest fragments and are generally poor at recolonizing disturbed areas. Over the past century, tropical rain forests have been reduced to about half of their original area. The rate of deforestation is currently accelerating, fueled by the resource demands of industrial nations and population pressures in developing countries. As biodiversity is a *public good*, it lacks assigned property rights, which in this case leads to market failure and resource over-exploitation.

Two Interpretations of Biodiversity and Their Implications for Conservation

In the *strict sense*, biodiversity refers to the "variety and variability among living organisms and the ecological complexes in which they occur." However, well-meaning attempts to link rain forest protection with sustainable development have led to a subtle expansion in the meaning of the phrase *biodiversity conservation*. In the *broad sense*, these words have come to represent everything from the maintenance of essential ecological processes and planetary life-support systems to the sustainable use natural resources. Ironically, this expanded definition of biodiversity conservation, with its focus on "green" development projects and saving "natural" lands, is only passively concerned with the avoidance of species extinctions or loss of genetic variability. This paper explores issues involved in the strict sense, for ultimately, if true biodiversity preservation is desired, direct preservation action will be required.

The Economist's Challenge: How to Value Biodiversity

The amount of biodiversity that will ultimately be preserved depends on its value to humans relative to other goods. In broad terms, rain forest biodiversity provides various products and services with market value, but then monocultures provide such goods as well. However, in the strict sense, biodiversity has two unique values, risk reduction and existence value. Diverse forests are less vulnerable to disease and catastrophic events; so, maintenance of biodiversity is valuable in that it *minimizes the risk* that the entire ecosystem will collapse, resulting in the loss of **all** goods and services. *Risk assessment* techniques can be used to determine the economic benefits of biodiversity's risk reduction value. *Existence value* is the amount of satisfaction that people derive from knowing that species exist in the world. The magnitude of this value can be derived with the *contingent valuation method* which uses survey techniques to create a hypothetical market in which respondents bid on their willingness to pay for the good in question.

The Ecologists's Challenge: How to Preserve Biodiversity Effectively

Protection efforts can be subdivided into direct and indirect types of conservation. *Direct biodiversity conservation* depends on maximizing the effectiveness of parks and reserves. Important factors include the overall *amount of land* to be protected, the *size and shape* of individual reserves (in general, large reserves are more stable and allow adaptive evolutionary change, while smaller reserves allow for the protection of scattered endemic species), and the *location* of the reserves (which considers the initial number of species and habitats). *Indirect means of biodiversity conservation* includes finding land-uses that simultaneously provide habitat to various species and economic returns to humans. This is necessary because no matter how elaborate and well-chosen a network of reserves may be (the *direct* method), most likely it will not be sufficient to maintain biodiversity at its present level. *Indirect* systems include *extractive reserves* [of non-timber forest products], *ecotourism reserves*, *selective timber extraction* [providing a sustained yield of wood], and *agroforestry* [serving as animal habitat and as a buffer zone.]

A Possible Solution: Integrated Land-Use Planning

The ecological and economic considerations discussed above imply that biodiversity conservation will be most effective in large-scale *integrated land-use complexes* that are in part funded by the developed world. These spatially integrated complexes combine direct and indirect biodiversity protection by coordinating land use over large areas. In general, such a system would have one or more *core reserve areas* surrounded by *economic activity zones*. Economic activities in the zones near the core area ideally would have little impact on diversity while those activities with potentially greater impact would be farther away. Corridors could possibly link complexes, enabling species to travel from one area to another to form *extended complexes*. Naturally, where human population density is very high and intact tropical rain forests have become islands in an altered landscape, the emphasis should be on maintaining essential ecological processes and services (broad biodiversity conservation), as the potential for strict biodiversity preservation is clearly reduced.

Establishment of these integrated complexes has several costs and benefits that accrue to different members of society. *Benefit-cost analysis* can be used to examine how establishment of different parts of such a complex affects various parties involved including local peoples, regional and national residents, and the global community. In general, establishment of core protected areas imposes an *opportunity cost* on local people and national governments who lose access to the resources contained within them. The *benefits*, however, largely accrue to residents of the developed world who generally have a higher existence value for biodiversity and who gain from tourism. Likewise, in surrounding economic activity zones, although some return does accrue to local people, they still must forgo alternative uses of the land while the benefits (reduced risk to the core area) still largely accrue to residents of the developed world. The implications of this benefit-cost analysis are that if the global community finds biodiversity conservation in tropical countries desirable, the global community must pay for it. Two potential sources of funding for large biodiversity protection projects include *debt for nature swaps* and the existing *Global Environmental Facility*. Whatever funding mechanism is used, projects should be designed to share control and financial responsibility with local peoples and governments.

Money that is raised from the developed world could be used to implement integrated land-use complexes. Necessary tasks include collecting baseline information to decide where to put parks, estimating demand for biodiversity conservation, developing land-use plans for selected areas, implementing these plans, and then enforcing the laws. Input and feedback from local peoples is necessary throughout the process.

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INTRODUCTION

Overview of the Biodiversity Crisis

How Many Species Exist?

In our solar system, only planet earth sustains life. It does so in such great splendor and diversity that no one knows how many kinds of organisms exist. Estimates vary from a conservative 3 to 5 million to a more extravagant 30 million *species* (May 1988), depending on how one extrapolates current knowledge. In particular, large numbers of insect species remain to be discovered (Erwin 1982). One recent, cautious estimate puts the total diversity at between 6 to 9 million species (Thomas 1990). Although scientists still debate this total number of species, they generally agree that the number of species had been increasing until very recently. The fossil record indicates that the earth's number of species reached a maximum at the end of the Pleistocene age, some 10,000 years ago (Niklas 1986).

What is Biodiversity?

The original definition of biodiversity refers to the "variety and variability among living organisms and the ecological complexes in which they occur" (U.S. Congress, Office of Technology Assessment, 1987). This *strict definition of biodiversity* is most commonly applied at the level of individual *species* (which are traditionally defined as a group of interbreeding individuals that are reproductively isolated from other such groups). It can, however, be extended downwards to cover genetic variability within a *population* (differences between individual organisms) and upwards to include *habitat and ecosystem diversity* (different types of communities). Here we focus on the species richness aspect, but genetic variability within species is also often affected by the eroding processes described below.

Where are Most Species Located?

Species richness is not evenly distributed over the surface of the earth. By all accounts, it is greatest in the tropics, reaching its peak in the evergreen forests located in moist to ever wet climates collectively known as *tropical rain forests* (see Box 1-1). Although tropical forests cover less than seven percent of the globe, they contain at least one half of all known biological species (Myers 1989; Wilson 1988; World Resources Institute 1987). Also, more than half of the as yet undiscovered species probably live in the tropics (Erwin 1982; Perry 1990). Ecologists agree that this curious concentration of species in the tropics is due to a combination of evolutionary and ecological forces. Over evolutionary time, for reasons that are poorly understood, there has been a higher rate of *speciation* (the formation of new species) in the tropics. Also, over ecological time, more species have been able to survive because they could subdivide more finely available *ecological niches* (a portion of the spectrum of resources within one habitat) and because greater differences between habitats could be maintained (Ricklefs 1987; Brown 1988).

Why are Rain Forest Species Vulnerable to Extinction?

Many processes contribute to the current threat to rain forest species. The most important of these are habitat loss, the fragmentation of remaining forests, the overexploitation of species in these fragments, and the introduction of new species. Species introductions are relatively unimportant in causing extinction in rain forests (except on deep-sea islands), so we shall concentrate on the first three of these factors.

World attention is currently focusing on tropical rain forests, not only because of their high biodiversity, but also because of their high vulnerability to human activities. Large-scale conversion of these forests began after World War II, following what has become a now predictable pattern of exploitation. This pattern of forest conversion first began to manifest itself in the temperate and subtropical zones some ten thousand years ago with the invention of agriculture (Prance 1991). A prime example of the pattern's effects can be seen in the Mediterranean region where large-scale logging and land-clearing for agriculture that occurred over two thousand years ago transformed extensive forest cover into the largely barren landscape existing today.

This pattern has since been repeated in many places and times including Europe during the Middle Ages, eastern North America since the late 17th century, and western North America during this century. In Europe, systematic logging has all but eliminated *primary forest* (Box 1-1), while in the eastern U.S. around two percent of the original forest remains, mostly in inaccessible sites (Postel and Ryan 1991).

In the temperate zone, the number of species pushed to extinction during these exploitation cycles has been relatively limited, consisting primarily of large mammals. Over evolutionary time, temperate zone communities have generally been subject to a high rate of large-scale natural disturbance (i.e. fires or glacial advances and retreats). Accordingly, many temperate plants and animals can cope with disturbance and are good at recolonizing newly regenerating habitats. A similar coping ability has evolved among species in tropical dry forest that are regularly subject to harsh seasonal climatic variations, extensive natural fires, or other forms of disturbance. Accordingly, although the dry forest has suffered an even greater reduction in area than the rain forest, so far relatively few species have gone extinct (Janzen 1986).

Tropical rain forests, by contrast, have been relatively stable over time. Although the Pleistocene glaciations have also affected their distribution, many refuges are thought to have persisted along rivers, in narrow valleys or at higher altitudes, so no long distance migrations of species were required. This relative stability has left the resident plants and animals less able to cope with extensive disturbance. For instance, a large proportion of birds and mammals of the rain forest have specialized feeding behavior and cannot survive in the drastically altered habitats resulting from human activities (Cranbrook 1988; Wells 1988). Likewise, mature-phase rain forest tree seedlings require specific sheltered microclimates provided by intact forest or by small canopy gaps in which to germinate and grow.

The past stability of the rain forests also has left resident species unable to recolonize converted lands. For example, even small belts of open vegetation may form formidable barriers to the dispersal of many rain forest animals (Diamond and May 1976). Since a high proportion of trees and vines in tropical rain forests rely on animals and birds for seed dispersal (Gentry 1982), the presence of these open areas can make the process of forest reestablishment painfully slow, if it occurs at all. Furthermore, regeneration in tropical forests growing on poor soils is often hampered by the loss of soil fertility that occurs when disturbance breaks the internal nutrient cycle, allowing critical nutrients to escape from the system.

Box 1-1:
Definitions of Forest Types

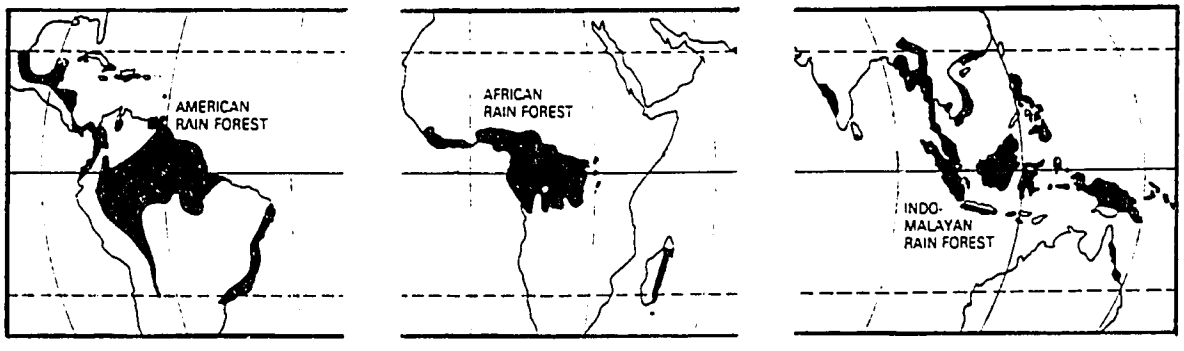
Tropical rain forests are forests occurring in those tropical climates (constantly high temperatures with mean temperature in the coldest month of at least 18 degrees Centigrade) that experience no or minor seasonal droughts (rainfall of at least 60 mm per month). These forests are mainly tall, evergreen, with a stratified and usually uneven canopy. They are rich in epiphytes and climbers, and have very high biodiversity. Since it is difficult to determine the elevational boundaries of this ecosystem, following Whitmore (1990), here we consider montane forests part of the tropical rain forest system. Figure 1 shows the original distribution of tropical rain forests.

Tropical dry forests are forests in a tropical climate growing in areas with highly seasonal rainfall, experiencing at least 3 months a year with severe droughts (less than 60 mm of rain per month). Compared to tropical rain forests, these forests tend to be shorter (except in gallery forest which grows in valley bottoms along permanent waterways), less stratified, and are dominated by deciduous trees. These forests are usually less diverse than rain forests although endemism (containing species that occur nowhere else) may be high.

Primary, virgin, or old growth forests are forests that have not been cleared on a large scale for at least several centuries. In this definition, scale is crucial, for natural dynamics of forests create many small openings (up to several hectares in massive windfalls after tropical storms). Slash-and-burn agriculture also produces temporary openings, which if sufficiently brief and small, theoretically leaves no permanent traces. A primary forest is characterized by complex age and size structure of the vegetation and high species diversity.

Secondary forests are forests that have been significantly affected by human activities. They are characterized by less complex vertical structure, lack of the larger size classes of trees, and a reduced biodiversity. Secondary forests cover abandoned agricultural areas or large-scale clear cuts. The rate of return from the secondary to the primary condition depends on the intensity, scale, and duration of the disturbance (Jordan 1986). The greater the change in soil characteristics and the larger the distance to sources of immigrants, the longer a forest will remain secondary.

Figure 1: Original Distribution of Tropical Rain Forests



Panels of cross sections of the globe

Habitat Loss in the Tropical Rain Forest Region

The tropical countries and regions with the greatest loss of natural rain forests are: most of Central America, the Caribbean, Colombia and Ecuador (west of the Andes), Southeastern Brazil, the whole of West Africa, Madagascar, most of South Asia, most of Indochina, and the Philippines.

Current rates of deforestation in the wet tropics vary dramatically from place to place (World Resources Institute 1991). The three countries with the largest portions of tropical rain forests--Brazil, Indonesia, and Zaïre--all have relatively low rates of deforestation. However, even within these countries there is much regional variation. For instance, the Indonesian island of Sumatra lost roughly one quarter of its forest cover just in the period 1980-1985.

The causes of deforestation vary as well. Globally, over half the destruction of tropical rain forests is due to largely uncontrolled conversion for agriculture, mostly for swidden farming (Myers 1989). Many of these cleared sites have poor soil and topography for permanent agriculture. Accordingly, many of the opened lands are soon abandoned. Increasingly, the people responsible for this are migrants from currently overpopulated areas. In some areas with low population density, logging is a major source of deforestation; in other areas, conversion rates are increased by (economically misguided) government subsidies to newly established cattle ranches.

What Factors Affect Local Extinction Rates?

The inability of resident organisms to adjust to converted tropical forest lands is one major cause of local extinctions. However, species are not necessarily safe in the forest fragments that remain (Diamond et al. 1987) (Box 1-2). First, the low population density of most species, which is a direct correlate of the overall high species diversity, makes it unlikely that *viable breeding populations* (sufficient numbers of individuals to avoid inbreeding or random extinction effects) will be contained in the fragment (Newmark 1991). Second, smaller fragments are unlikely to include the optimal habitat or the required habitat mosaic of many highly specialized species commonly found in tropical communities (Karr 1982). Third, many forest animals make irregular or seasonal treks into other forest types when their normal habitat does not provide enough food. Obviously, forest fragments lacking the appropriate mix of habitats will eventually lose many of those species. Finally, there may be a strong *edge effect* (increase in disturbance at the edge of the forest) in smaller fragments. For instance, birds in small forest patches experience increased rates of nest predation and brood parasitism (Wilcove and Robinson 1990; Bierregaard 1990).

The rate of local extinction in forest fragments also increases because of the interdependence of species in the ecosystem. Once certain species are eliminated, others are likely to follow in a cascade of secondary extinctions. For instance, some plants rely on a very limited set of bird or mammal species for their dispersal (Beehler 1988). The extinction of the dispersers will inevitably reduce the prospects for long-term survival of those plants, and of the species immediately relying on them for their livelihood. A second example is that large predators, which are quite vulnerable to extinction due to fragmentation, help to maintain the diversity of their prey species. Some of those prey, in turn, destroy the seeds of trees in a highly selective fashion, which may help to maintain local tree diversity (Terborgh 1988). Such gradual coming apart of the ecological community is being witnessed now in smaller fragments (Dirzo and Miranda 1990). In a few more decades, when equilibrium is reached in remaining reserves, we may see species richness impoverished in all but the largest fragments unless the populations of the most vulnerable species are actively managed.

Box 1-2:
Deforestation and Biodiversity

The number of species found within a given area depends on the diversity of its abiotic features (soil types, altitudes, etc.), its size, and its isolation. In most ecosystems, the relationship between size and diversity is usually predictable because any smaller area is a sample from a larger one and thus unlikely to contain all species. The curve is often best described by a linear log-log plot, with a slope of around 0.3. However, steeper slopes and lower numbers at small sizes are often found to characterize island situations where recolonization from a large pool of species (a mainland or very large island) is difficult. This *island biogeography model*, when applied to tropical deforestation, predicts two effects: (1) an immediate effect of a reduction in biodiversity because of size reduction, and (2) a long-term effect of a reduction in biodiversity due to a lack of replacement of populations that went extinct in the forest fragment.

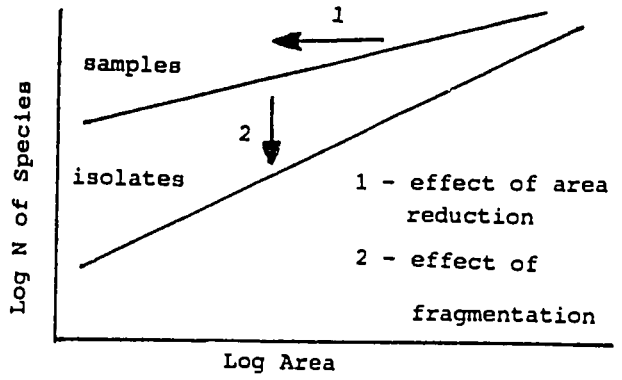


Figure 2

The magnitude of these effects is unclear. This is because the extent to which the analogy of real islands and their underlying processes is applicable to habitat fragments, is unknown. The applicability of this model has been criticized for several reasons (Mann 1991). First, it has been argued that the species-area curve levels off as areas become very large. However, careful inspection of available species-area curves provides little basis for such a conclusion (Shafer 1990). Second, as mentioned above, the island analogy may not be appropriate. Where forest fragments are surrounded by secondary woody vegetation or heavily exploited forests rather than agricultural fields or pastures, this vegetation may be less of a barrier than suggested by the island analogy, at least for a subset of the forest species. Indeed, a recent study of Australian rain forest mammals showed that survival in fragments of primary forests depended mostly on the ability to live in the modified habitats surrounding the fragments (Laurance 1991). As would be expected, deep-forests specialists are most likely to go extinct. Ultimately, research is showing that fragmentation effects are undeniable, even if they perhaps do not follow the quantitative prediction of the species-area curve for isolated islands, they are undeniable.

Overexploitation

Even if a forest is not converted it may lose species due to incessant exploitation by humans, or even by mere human traffic. Fragmentation or penetration by roads constructed by governments or logging companies expose forests to exploitation. In much of the American tropics (the neotropics) and West Africa many forests have lost most or all large game species due to relentless hunting pressure (Robinson and Redford 1991; Redford 1992). Plants are also sometimes overharvested to the point of local depletion (e.g. gaharu or rattans in Southeast Asian forests; van Schaik, pers. obs.). Selective logging often leads to loss of plant and animal species of the primary (old growth) forest due to excessive damage to the remaining stand (Johns 1988, 1991). Intense human traffic through rain forests, even if restricted to foot trails, may lead to reduced diversity of wildlife and changes in the community structure (Griffiths and van Schaik in press). Many rain forest organisms, be they plants or animals, have such slow life histories that they are much more vulnerable to harvesting than those of savannas or temperate regions (Robinson and Redford 1991).

While in many forests most of these activities have taken place for hundreds or thousands of years, they are now occurring at unprecedented scale and intensity. Increased population pressure, increased penetration of hunters and collectors, improved technology (firearms vs. bows and arrows), and above all, commercialization (rather than subsistence use) are the main reasons for loss of sustainability.

From Local Extinctions to Loss of Species

The combined effects of habitat loss, fragmentation and overexploitation, lead to reduced local species richness (Box 1-2). Clearly, extinction of many local populations of a species will eventually lead to the demise of the species itself. The risk of this happening is directly dependent on the size of the species distribution area. Since local endemism of plants and animals is much more common in the tropics than in the temperate zone (Gentry 1986; Terborgh and Winter 1980; Pagel et al. 1991), loss of area and habitat fragmentation will inevitably lead to species extinctions.

Several species extinctions are now being recorded. Scores of species of plant and an unknown number of animals were lost in Western Ecuador when most of the forest containing a high proportion of narrow endemics was lost in the early 1980s (Gentry 1986). Similar examples are now being documented throughout the world (Prance 1991; Thiollay and Meyburg 1988).

Usually, however, we must estimate extinction rates, simply because no inventories exist for most of the forests that are disappearing. These estimates are based on known species-area relations for some well-studied groups of species (Box 1-2), on rates of deforestation, and on estimated total number of biological species. For instance Simberloff (1986) estimated that when the area of rain forests in the Amazon becomes half its original size, approximately 15% of all plant and animal species will be lost. If the Amazon loses all forests outside current reserves, some 70% of species will be lost.

Estimates of the actual number of species lost are critically dependent on the number of species thought to be present before the area loss. In 1987, Wilson estimated that each year some 10,000 species are going extinct due to tropical deforestation; his current estimate is 100,000 per year (Mann 1991). These estimates have been criticized as overly alarmist (Box 1-2). Although actual values may be much lower (or much higher!), it is difficult to avoid the conclusion that substantial loss of species is imminent.

If current trends in deforestation remain constant, extrapolations make various predictions that present rain forest area will be halved in another 20 to 30 years, and that by the year 2050, both tropical rain forests and their species will have disappeared. Unless action is taken to change the deforestation process, we are likely to witness the largest spasm of species extinctions in the history of our planet. During the long history of evolution, species extinctions have always occurred, sometimes even in dramatic waves. However, these declines took thousands of years, and their impact was undone within a few million years. This time the rate of loss is far higher than it was even during those times of evolutionary upheaval (Myers 1989).

Two Interpretations of Biodiversity

Biodiversity in the Strict Sense

So far we have mainly referred to the species richness component of biodiversity. Biodiversity also has an intraspecific (genetic) component and a supraspecific (community-level or landscape-level) component. All these components, however, have in common that they refer to genetic variability, both within and among biological species. We shall refer to this definition as the strict-sense definition of biodiversity.

Biodiversity in the Broad Sense

Over the past few years, however, a new, less specific perception of biodiversity has developed among both the conservation and development community and the general public. This perception can be traced to the World Conservation Strategy, which emphasizes the maintenance of essential ecological processes and life-support systems and sustainable use of resources in addition to the classic focus on biological diversity itself. In effect, this *broad definition of biodiversity* has become a symbol or code word for biological resources. In this broad sense, the value of biodiversity thus is defined as the total value of products (consumptive and non-consumptive uses) and services provided by natural ecosystems and living things. For example, McNeely et al. in their influential report, "Conserving the World's Biodiversity" (1990), use the fact that "firewood and dung provide over 90 percent of the primary energy needs in Nepal" as an illustration of the values of biodiversity. Gradually, biodiversity conservation has come to mean not just the preservation of the variety and variability of living things, but any type of "green" development project ranging from the establishment of national parks to the development of agroforestry systems for fuel wood production.

The Importance of Biodiversity in the Strict Sense

The conceptual distinction between the strict and broad definitions of biodiversity is significant, and not just a semantic argument. Furthermore, we are most emphatically <u>not</u> condemning the intentions or objectives lying behind the use of the broader definition of biodiversity. Indeed, it is generally agreed that development is critically dependent upon the maintenance of ecological processes and the sustainable use of self-renewing natural resources. The broader concept of biodiversity has become a powerful tool by which to pursue conservation and sustainable use objectives by maintaining natural areas for the many valuable products and services that they provide.

Our point is that by focusing on biodiversity in the broad sense with all its attendant human development issues, we might forget the fundamental problem of the biodiversity crisis: the world is beginning to lose *species* at a rate unprecedented in the history of the earth. The vast and urgent nature of the biodiversity crisis makes it extremely unlikely that the current emphasis on broad biodiversity development projects will prevent catastrophic extinctions, no matter how successful these projects are in maintaining 'natural' lands. *If biodiversity preservation is desired, biodiversity preserving action will be required.* In the rest of this paper we will use biodiversity in the strict sense.

Economic Explanations for Habitat Loss and Species Extinction

Two main economic factors seem to be driving the rapid destruction of tropical forest and associated species extinctions: 1) the high opportunity cost of land in alternative uses, and 2) market failures.

The *high opportunity cost* of tropical forest lands (the value of alternative land-uses such as agriculture or timber) is directly linked to population pressure in host countries. For example, from 1975 to 1986, 2.8 million hectares of forested land were cleared in the Brazilian state of Rondonia alone, mostly by small farmers and cattle ranchers, with the active encouragement of the Brazilian government (Miller et al. 1991). This suggests that the opportunity costs of land in forests are significant, and that the net benefits from using land in alternative uses are higher than those from leaving land in forests. In some cases these high opportunity costs can be attributed to government-provided subsidies for agricultural uses of land. However, many other important factors help explain the loss of major tracts of forest land.

Various types of *market failure* (the breakdown of economic efficiency caused by conditions that violate the assumptions of microeconomic theory) also cause the loss of biodiversity (McNeely 1988). One such failure is due to the *open access nature* (lack of defined property rights) of forest resources and services, which contributes to the degradation of many different biological resources. For instance, in India during British colonial rule, there was a disruption of the communal organizations that controlled land use. This resulted in the conversion of communally managed resources into open access systems which were then exhausted (Gadgil and Iyer 1989). A current example of a country with open access problems is Madagascar. Here shifting cultivation is practiced on approximately 200,000 hectares of public lands every year. This cultivation, which was sustainable when the population was small, is one of the prime reasons for the rapid deforestation of Madagascar's eastern rain forests (Andraimpianina 1985).

There are two additional aspects of market failure that lead to tropical deforestation and loss of biodiversity. One is that biological goods are *public goods:* goods and services which are consumed jointly by a number of different individuals and for which it is extremely difficult to attribute the costs of consumption to any specific person or group. Thus, there is a lack of accountability with public goods that allows for their depletion. The second aspect, related to the first, is the *undervaluation* of biodiversity. Undervaluation occurs for goods which have no markets, and consequently, no prices. For most natural resources, such as oil or copper--for which markets and prices exist--complete exhaustion will not occur; as resources become more scarce (usually reflected by rising prices), people respond by conserving the remainder or switching to substitutes. With tropical forests, however, because the goods and services provided by biodiversity are not priced, the costs of preservation outweigh the benefits, and the resource is used to exhaustion. Thus, significant changes in government polices are needed to halt overuse.

THE ECONOMIST'S CHALLENGE: HOW TO VALUE BIODIVERSITY

Intuitively, the most effective measure to preserve biodiversity would be to stop all human impacts on the world's remaining natural areas. In a world of limited resources, however, this extreme action clearly is not a viable option. Ultimately, the amount of biodiversity that will be preserved depends on its value to humans relative to other allocations of resources.

Although there are many non-anthropocentric ethical arguments for saving species from extinction (Box 2-1), the people making such arguments are really saying that they assign a very high value to biodiversity. A fundamental step in the management and preservation of biodiversity thus is determining the value that a given society or humanity in general assigns to various levels of biodiversity preservation.

From the traditional economic point of view, the value of biodiversity is derived from the value of goods and services provided to humans, now and in the future (Randall, 1991). Discussions about the value of biodiversity in the literature often focus on the consumptive or use value to humans arising from the flow of outputs from a diverse forest. Such outputs include fuel wood, fruits, nuts, traditional medicines (McNeely 1988), ecotourism, and potential new products or compounds with pharmaceutical applications (Principe 1991). Here, by contrast, we are concerned with estimating the value of biodiversity in the strict sense. Thus, we do not consider consumptive or use values. Rather, we focus on those values that depend on biodiversity per se: (i) the reduced risk of major upheavals in the flow of goods and services, and (ii) existence values.

Risk Reduction

A standard broad biodiversity argument for preserving forested areas is the value of the ecological products and services they provide such as timber and non-timber forest products, regulation of downstream water supplies, stabilization of local climates, improvement of air quality, protection against erosion, provision of pollinators, dispersers and pest control agents for nearby crops, and removal of carbon dioxide from the atmosphere.

From a strict biodiversity perspective, however, one could argue that by themselves, none of these services are a direct result of biodiversity--indeed provision of any one of these services could probably be maintained, or even increased, if the system is managed, and simplified biologically. However, such a simplification increases the vulnerability of the system to severe, sometimes catastrophic, reductions in the system's abilities to provide these essential services, let alone other, equally essential ones.

A useful analogy can be made between biodiversity management and financial management. Portfolio theory has long been used to analyze and improve the management of financial assets. Although investors might have great confidence in the ability of a particular stock to increase at a rapid rate over time, they would be unwise to invest all of their personal funds in that single stock because of the significant risk that the stock might actually fall in value due to unforeseen circumstances. A prudent investment manager diversifies his or her holding to reduce the risk of his or her assets falling due to downturns for a particular firm or sector of the economy. An investor holding a diversified portfolio may well earn a lower rate of return over five years than an investor holding a less diversified portfolio. Nonetheless, many investors are willing to give up at least a modest amount of expected return in order to avoid large fluctuations in their overall asset values.

Box 2-1:
Ethical Arguments Underlying the Economics of Biodiversity Preservation

The issue of biodiversity preservation extends beyond the realms of ecology and economics to pose deep moral and ethical questions. There is a wide range of ethical arguments made, not so much for and against preservation, but about the rationale for preservation and the appropriate level of preservation. Some argue that biological resources are beyond value since they provide biotic material which underpin every major type of economic endeavor. This is the ultimate instrumentalist argument. Others argue that human destruction of biodiversity is indefensible, because non-human species also have rights to resources which are currently being monopolized or over exploited by humans. This raises the question of fairness in distribution of access to resources, across species. Closely related to the above point of view is the argument that all life forms are intrinsically valuable and have a right to exist (Callicot 1986). Therefore, by careless destruction of biodiversity, humans are failing to acknowledge this fundamental right.

Biodiversity is treated by economists as a scarce resource. For the adherents of neoclassical economics, the value for any good or service, including biodiversity, emerges from individual preference rankings within a constrained environment. There is a strong normative basis for this argument. Individuals are seen as being the best judges of what is good for themselves (Randall 1986). Therefore the "value" of different goods and services depends on the choices made by individuals. These choices are presumed to have been made in a thoughtful and careful manner. This is clearly based on the utilitarian ethic, but this approach also acknowledges that a major, if not dominant, component of the value people assign to biodiversity will be for nonuse reasons. For example, individuals may be willing to pay to preserve biodiversity habitat even though they may not expect to get any direct benefit from the preservation other than the satisfaction of having contributed to what they perceive to be the "right thing to do."

A similar argument can thus be made for natural forest ecosystems. Natural forests have evolved into stable ecosystems that can survive without human interference. A forest cleared for a pine plantation may maximize wood production and watershed management, but such a plantation becomes vulnerable to factors ranging from disease invasion to a crash in the world price for softwood. These catastrophic events could then cause the managed ecosystem to crash. In natural forests, by contrast, the system minimizes the risk of invasion by disease. Furthermore, the system is not dependent on just one benefit but on several different benefits.

One approach to developing strategies for biodiversity protection which explicitly accounts for risk is the *safe minimum standard* strategy (SMS). Bishop (1978) argues that this is an appropriate way to preserve a species at a level sufficient to ensure its survival. He suggests the adoption of strategies which minimize the maximum possible loss (minimax strategy).

While Bishop uses this approach for preserving a single species, we can generalize it to a level that aims at preventing the collapse of entire ecosystems. Such an approach supports the view that we must conserve biodiversity unless there are larger net benefits associated with development. In the conservation versus development debate, this shifts the burden of proof to those proposing development (Tisdale 1990).

Existence Value

Can Economics Measure Non-Monetary Values?

A second set of economic values arising from strict biodiversity are what are commonly called intangible values. Probably the most important intangible value in this context is existence value. Many people derive satisfaction from knowing that elephants exist in the wild even though they will never directly observe them. Similarly, many people appear to value the diversity of species associated with tropical

forests and other diverse ecosystems. This value is shown by the large amount of voluntary contributions to organizations such as the World Wildlife Fund and Rainforest Action Network.

A rough measure of these existence values for a particular area would be the total contribution by foreign entities to biodiversity projects in the area. For example, under the Madagascar Environmental Action Plan, a variety of donors committed to spending approximately $70 million on biodiversity protection and related activities (World Bank 1990). Of course there are several limitations to this crude approach to estimating existence values associated with biodiversity. First, given the current political popularity of biodiversity protection, many things may be included in biodiversity protection projects that have little or nothing to do with strict biodiversity. Second, there is at best a loose link between the biodiversity funding decisions of foreign development agencies and the interest in biodiversity among the citizens of the donor country.

A more direct approach to estimating existence values is the *contingent valuation method*. Contingent valuation is a method which combines survey research methods with economic theory to obtain estimates of a change in the level of provision of a public good, such as clean air. Surveys are constructed which create an appropriate hypothetical market in which individuals have an opportunity to bid on the good in question. The method relies heavily on cognitive psychology and sample survey methods.

The contingent valuation method is so named because the elicited bids are contingent (depend) on the particular market described to the respondents. One can construct hypothetical markets to determine willingness to pay for reductions in air and water pollution, prevented declines in biodiversity, improved management of hazardous wastes, improved recreational opportunities, or for any number of goods. Economists have used the contingent valuation method to value endangered species such as humpback whales, whooping cranes, and bald eagles (Loomis and Walsh 1986). Of course, valuing a single, often well-known endangered species is much different than valuing the far more complex good known as strict biodiversity. Nevertheless, this economic valuation work on endangered species is a good starting point for understanding the challenges facing economic analysts interested in biodiversity.

Research on valuing the protection of endangered species shows that existence values are much larger than consumptive or use values. People are willing to pay positive amounts of money for programs to protect endangered species, not because they expect to have any direct contact with them, but because they derive satisfaction from knowing they will continue to exist in the wild (Box 2-2). Another finding is that people are willing to pay more for species with which they are more familiar. This is consistent with the findings on contingent valuation of other environmental resources. This implies that biodiversity values will depend on how knowledgeable people are about biodiversity, and also suggests that value estimates obtained in empirical studies may be influenced by the amount and type of information about biodiversity conveyed to people during the course of the survey. Another finding is that species that are "cute and cuddly" (species like chimpanzees, for instance, that endear themselves to people) get higher assigned values than those that are not. For valuing strict biodiversity, this implies that forests with identical diversity ratings may receive quite different values from survey respondents depending on which species are contained in the ecosystem.

Existence Values in Developed and Less Developed Countries

Up to this point, the discussion of existence values has ignored the role of income in determining the value of biodiversity. In fact, income plays a critical role. In general, the demand for environmental services is *income elastic* (the increase in quantity demanded is greater than the per unit increase in income). In higher income countries worldwide, we see evidence of this in the growing demand for more

Box 2-2:
Using the Contingent Valuation Method (CVM) to Value Wildlife Preservation

One example of how CVM can be used to value wildlife preservation benefits is a study by Walsh, Bjonback, Rosenthal, and Aiken (1986). In this study, the benefits of wildlife preservation were estimated for the residents of the state of Colorado. The basic data were obtained from a survey of 198 randomly selected households. The procedure adopted was to first define the simulated market by describing the existing endangered wildlife programs in Colorado. This was followed by descriptions of two possible alternative states; first, a deteriorated state resulting from removing all wildlife programs, and second, an enhanced state with improved wildlife programs. (Visual aids were used to reinforce the verbal descriptions of the two alternative states.) Respondents were than asked to report their maximum willingness to pay for the wildlife programs that would prevent the deteriorated state, and would attain the enhanced state. Respondents were also asked to allocate the amount reported among use and non-use categories of value. The results obtained in this study indicate, that on average each household is willing to pay $58/year for recreation use and wildlife preservation in Colorado. This implies that the benefits from wildlife preservation in this state annually amount to $68.9 million. Approximately 70% of this amount can be attributed to the non-use benefits people derive from preservation.

parks and reserves, stricter water and air pollution regulations, and so-called "environmentally friendly" products. However, in lower income countries, where many people are still living at a subsistence level, environmental concerns are less prominent. For those interested in designing programs and policies to maintain biodiversity, this creates a dilemma: most of the benefits (measured by demand) are going to people in countries other than where much of the biodiversity is found. Furthermore, most of the costs of protecting biodiversity seem to fall on those in developing countries where the predominant concern is meeting basic human needs. This implies that maintenance of biodiversity is a global environmental good; protecting biodiversity in the tropics provides benefits to people on all continents. Thus, international cooperation is necessary to charge the beneficiaries for what they receive. Some innovative means of financing biodiversity protection measures are discussed in the last chapter.

THE ECOLOGIST'S CHALLENGE: HOW TO EFFICIENTLY ALLOCATE BIODIVERSITY-PRESERVING RESOURCES

Ultimately, the value that a society places on biodiversity is a measurement of the amount of resources that it is willing to devote to biodiversity preservation. Once economists have estimated this value, ecologists then must determine how to make use of these resources most efficiently. In other words, ecologists must decide how to get the most "diversity for the dollar."

There are two basic approaches to the maintenance of diversity, either direct or indirect means. Direct biodiversity preservation focuses on establishing reserves whose sole purpose is the preservation of natural ecosystems. Indirect biodiversity preservation, on the other hand, involves developing land-use systems that preserve some level of biodiversity while simultaneously providing other economic returns. The resources allocated by a given society for biodiversity preservation will be "spent" in some combination of direct and indirect approaches. Because of the trade-off between these two methods, we need an understanding of the ecological issues behind each method to reach an efficient resource allocation.

Box 3-1:
Geographic Patterns in the Biodiversity of Tropical Rain Forests.

The world can conveniently be divided into biogeographic regions. Within each region flora and fauna show great similarity. Tropical rain forests occur in all major tropical realms: the Neotropical, Afrotropical, Indomalayan, Oceanian, and Australian realm (see Figure 1 in Box 1-1). Each realm is divided into biogeographic provinces. The International Union for the Conservation of Nature (IUCN) has made use of these provinces as the natural units for the organization of a network of protected areas. Each province should have at least one major protected natural system. Where provincial boundaries do not coincide with political boundaries, countries are expected to protect the biogeographic provinces within their borders.

The rain forest biomes in the different realms vary in overall diversity. This diversity is due to both the size of the biome and to climatic variability over time. The South American forests are in the largest contiguous blocks (compared to Asia and Africa), have experienced the effects of Pleistocene droughts, and have the highest number of species. Southeast Asian forests are broken up into many smaller units, were relatively little affected by the Pleistocene droughts, and have an intermediate amount of biological diversity. African forests were most affected by the Pleistocene droughts, shrank the most, and are generally less diverse. It should be emphasized, however, that these realms show virtually no overlap in their biological diversity, all being uniquely different.

Within provinces, biodiversity sometimes varies predictably in relation to the Pleistocene history. For instance, in South America, some two dozen areas with high biodiversity and high incidence of endemism (occurrence of species occurring only in one limited geographic region) show reasonable coincidence with the presumed locations of the Pleistocene refuges into which the forest withdrew during the height of the glaciations (Whitmore 1990). In Africa, three such areas have been recognized. Although the theory that the events during the Pleistocene shaped the biogeography of Neotropical and Afrotropical rain forest organisms is still controversial, it forms the basis for the planning of reserves in Africa and South America. No such refugia-related biodiversity patterns are apparent in Southeast Asia, where island size is a more important determinant of biodiversity.

Reserve Design and Direct Biodiversity Conservation

Considering only direct biodiversity preservation, the critical question in reserve design is: How much biodiversity preservation do we get from a given amount of allocated reserve land? Answers to this question depend on the relative size, actual area, and location of the protected areas.

What Percent of an Area is Optimal for a Reserve?

The first step in the design of reserves is to decide what percentage of land area should be protected. In other words, in a given area, how much land should be protected relative to the amount unprotected? Ecologists have subdivided the world into biogeographical provinces, each of which contain unique assemblages of fauna and flora (Box 3-1). Within a given province, successful biodiversity conservation depends on maintaining a sufficient representation of regional species and habitats.

In general, a fairly predictable positive relationship exists between area of a given piece of land and the number of species it contains (Box 1-2). This relationship occurs because as the land-area of a reserve increases. the number of habitats it contains also increases. Accordingly, if all of the land-area in a given province were to be preserved in its natural condition, then (at least over the short-term) all of the species present would be maintained. If, however, only 50% of the land-area were protected and the rest converted into a biologically barren landscape, we could expect to maintain only around 80% of the species originally occurring in the region (Shafer 1990). And if only 10-20% of the land-area were preserved, then perhaps as few as 50% of the species would survive. The implication is that the most

Box 3-2
Preservation of Biodiversity: In situ or Ex situ?

The difficulty of protecting intact ecosystems against illegal encroachment and degradation by climatic changes has led some to argue strongly for maintaining stocks of species that are threatened and/or potentially important economically in controlled conditions outside reserves. However laudable the objectives, this approach is likely to pay off only for species of proven economic importance. Ex situ (off-site) conservation can never replace in situ (on-site) conservation for the great majority of species for three reasons. First, there are simply too many threatened and economically valuable species. (Moreover, new species are being identified every day.) Second, it is impractical to collect genetic material of all these species without substantial loss of genetic variation and thus potential for evolutionary change. Many of the collections drawn up in the past have suffered appreciable losses (Whitmore 1990). Third, species can effectively change and adapt to new parasites and pathogens only in their natural environment.

If a species is threatened with extinction because of direct human exploitation (hunting, poaching), ex situ protection may be effective. However, these are often big species that play an important ecological role, and that role would be lost when they are removed from their environment.

politically viable and likely outcome is that 90% of important biogeographic areas will be exploited and 50% of the species will be protected.

The exact values of these estimates are still uncertain because they are based on studies of real islands. The experimental reduction in area to test the validity of this island model to habitat islands on the mainland is only now taking place (Box 1-2). Currently, only 3-6% of the world's tropical rain forest region is formally protected in reserves (World Resources Institute 1991). Although this percentage varies between countries and may be increased in the future, it is unlikely to go much beyond 10%, unless the value of biodiversity makes such an increase economically sensible. Thus, if protected areas are the only good habitats available for rain forest species, an appreciable loss of species seems inevitable, especially since ex situ conservation is an option for only a tiny minority of these species (Box 3-2).

What is an Optimal Reserve Size?

The second step in design of reserves is to determine how big each reserve should be. When a long-term time horizon is considered, the equilibrium number of species maintained within a given reserve depends not just on the variety of habitats it contains, but also on its size (Simberloff 1986). Given a fixed amount of land area that can be allocated to reserves within a biogeographical province, it is necessary to decide whether it is preferable to have a few large reserves or many smaller ones.

On one hand, smaller reserves make it possible to protect many different sites. This large number of sites ensures that the most unusual habitats and rare endemic species, which occur only within one region and nowhere else, are represented. Moreover, many of the ecological services of the forest are fairly localized in effect, making a network of many smaller areas preferable from the point of view of broad biodiversity protection.

On the other hand, large reserves are more likely to preserve species over the long-term. As outlined earlier, size is a major factor in determining whether a reserve will protect viable populations of rarer species, some of which can play a major ecological role in maintaining the overall ecosystem. Furthermore, since larger reserves generally contain a wider diversity of habitats and physical gradients (rainfall, altitude, etc), the biological communities in these reserves will be better able to adjust their local distributions in response to climatic or other environmental changes, such as those predicted as a result

Box 3-3:
Local Biodiversity Maxima

Within a single region, biodiversity follows some predictable patterns. Planning of protected areas can make use of these patterns to identify the areas most in need of protection, or most likely to have high diversity or higher densities of extinction-prone species. However, this does not mean that other areas are unimportant. Although the organisms in different habitats overlap in species composition, each habitat has some or many distinctive species characteristic for that habitat only.

Lowland forests are richest in species for three reasons. Diversity generally declines with altitude, although the organisms inhabiting the higher elevations are not a mere subset of the plants and animals living in the lowlands (Cranbrook 1988; Wells 1988; Whitmore 1990). Secondly, forests on fertile soils often contain more diverse communities than those on poor soils (Gentry and Emmons 1984; Emmons 1984), although, again, forests on different soils also tend to have characteristic plant and animal species. A final factor is rainfall. Within South America it has been established that local plant diversity increases with rainfall up to about 4,000 mm annual rainfall (Gentry 1982). Therefore, lowland forests on fertile soils are the richest ecosystems on earth. Because of their obvious suitability for permanent agriculture, they are also the most threatened ecosystems and, as pointed out earlier, are unlikely to provide enough direct-use environmental services to outweigh the appreciable direct economic gains of transforming them into agricultural land.

of the greenhouse effect. In general, since biodiversity preservation necessarily has an infinite time planning horizon, the long-term stability and resilience of the large reserves would make them the best option for the preservation of biodiversity.

What is an Optimal Location for Reserves?

The third step in reserve design is to carefully locate reserves in areas of outstanding local biodiversity so as to increase the proportion of species protected per unit of land area. For instance, if only one reserve is planned in a region, then biodiversity preservation will be maximized by placing it in a location with a high diversity of habitats. In the absence of precise information, one can make use of some predictable patterns (Box 3-3). If two reserves are planned, then biodiversity preservation will be maximized by placing them in two different habitat-rich ecosystems. This process of adding reserves should continue until the benefit gained in species preservation from adding an additional reserve is equal to the cost of establishing the reserve. In economic terms, the optimal allocation of land to reserves is (all other factors being equal) where the marginal value of increased species preservation is equal to the marginal cost of the best alternative land use.

Existing reserves may not be optimal in many cases. For instance, in some tropical countries, these networks date back to the colonial era, when knowledge of biodiversity patterns was relatively limited, and conservation was very utilitarian. Accordingly, areas of high agricultural potential were excluded from many reserves. Today, we find maximum diversity in habitats that are under the most pressure to be converted to agricultural uses: lowland rain forests on the most fertile, usually alluvial or volcanic, soils. Therefore, a top priority is to make some of these habitat patches part of larger "conservation complexes." A review of the protected area networks of several countries is needed to incorporate some of these extremely rich patches into existing reserves.

Indirect Biodiversity Preservation via Biodiversity Compatible Land-Uses

As mentioned earlier the most effective step that could be taken to preserve biodiversity would be to stop all human impact on the world's remaining natural areas. However, a figure of around 10% of the land surface in tropical countries set aside for conservation may be more politically realistic. Given the resulting loss of species, it is clear that areas outside reserves must be managed so as to continue to contribute to biodiversity maintenance (Salwasser 1987).

Direct biodiversity preservation in parks and reserves can be supplemented with indirect biodiversity preservation in lands being used for other purposes. For instance, people are trying land-use alternatives that help maintain biodiversity while also giving economic returns. It may be possible to design exploitation methods (i.e. hunting, settlement patterns) that have less impact on local biodiversity and vegetation.

Extractive Reserves

Much attention is currently focused on extractive reserves that emphasize the sustainable exploitation of the non-timber resources. Such reserves obviously form a valuable strategy for preserving biodiversity since the natural system is modified very little, provided the harvesting of the extracted resources is done sustainably. In a limited range of situations, these reserves form the optimal kind of multiple land use. However, it is not a universal panacea, because, in many situations, extractive reserves are economically less viable than commonly assumed (Browder 1992; Salafsky et al. in press) and in some cases, the sustainability of the extraction is unlikely. Hunting, while traditionally an important source of meat for local communities (e.g. Caldecott 1988), must be managed very carefully if it is not to lead to local extinction of all larger birds and mammals of the rain forest (Redford 1992). It is an unfortunate coincidence that many of the species vulnerable to hunting tend to play an important ecological role in their communities, as predators, dispersers, or modifiers of vegetation structure.

Ecotourism

Ecotourism can be both economically profitable and biologically sustainable. The ecotourism industry as it exists is mainly driven by demand from tourists from the developed world. For example, in Rwanda, "gorilla tourism" brings in roughly $1 million per year directly through entrance fees and about $9 million indirectly; Kenya is estimated to have earned $500 million from nature tourism in 1988 (Lindberg 1991); Ecuador receives some 260,000 international visitors annually and a recent study by WWF shows that 76% of a sample of tourists interviewed were mainly ecotourists (Boo 1990).

As the ecotourism industry results in foreign exchange earnings for tropical countries, governments of these countries have an incentive to protect nature. However, ecotourism must compete for governmental support with two traditional industries which are also foreign exchange earners--the timber industry and cattle farming--which historically have contributed considerably to deforestation (see section 1.1). This, and the competition between different tropical forest based tourism sites are potential constraints to the growth of the ecotourism industry. However, if demand continues to increase for ecotourism, these two constraints may become less significant.

More recently the ecotourism industry has been expanding to include domestic tourists. This is a good sign for biodiversity protection, in that political factors stirred by national sentiment for protection may act as a hindrance to deforestation. However, so long as average income levels are low in countries where biodiversity protection is most needed, domestic ecotourism, a highly income elastic good, is likely to be limited.

An important bottleneck to the development of ecotourism is the lack of infrastructure within many of the countries under consideration (Laarman and Durst). This is a dual-edged problem. The lack of infrastructure within the range of protected areas can be modified by including development of infrastructure into plans for protecting a given ecosystem. However, in doing so there is the potential danger that the increased demand which infrastructure development promotes will result in environmental damage, congestion and cultural disruption (Lindberg 1991).

Selective Timber Extraction

Logging is likely to remain profitable in virtually every region of the tropical rain forest biome in light of the expected increase in the demand for tropical hardwoods (Lean et al. 1990). A great variety of selective logging procedures exist that do not lead to reduced timber yields in the long run (Jonsson and Lindgren 1990). They vary in the extent to which management modifies the structure and composition of the natural forest (Putz 1991). The systems that are best at mimicking the natural dynamics of a forest and minimize the attendant damage to the parts of the forest that are not being harvested are most likely to be compatible with biodiversity. The few available ecological studies indicate that selective logging, if conducted properly, need not lead to drastic losses of biodiversity (Johns 1988, 1991), although some species remain vulnerable to even the most careful timber extraction systems.

Tropical forestry has long been a mining industry. As recently as the late 1980s, less than 1% of the tropical forests were harvested in a sustainable manner for their timber (Poore et al. 1989). Given the rapidly diminishing supply of accessible unconverted rain forests and the concomitant decline in tropical hardwood production, the urgent development of sustainable forestry is dictated as much by economic factors as by ecological considerations. The challenge is to develop systems that are least damaging to the ecological processes of the forest ecosystem and its biodiversity while at the same time yielding significant economic returns. Little systematic ecological research has been conducted to evaluate the consequences for biodiversity of different sustainable logging systems.

In the past, logging concessions often did not get a chance to develop to their second cut because the roads made by the concessionaires opened up the land, and the logging made it easy for colonists to establish swidden plots. Also, commercial hunting is very common in those concessions that were not converted into agricultural land. For these reasons, sustainable forestry is least likely to succeed where population density is high (Putz 1991). In such areas agroforestry or community based forest plantations remain the only options.

Agroforestry and Plantations

Agroforestry and (mixed-species) plantation forestry are forms of land use that are less damaging to biodiversity than other forms of agriculture. However, agroforestry in particular is linked to high population density and intensive land use, and most appropriately developed in the zones surrounding protected areas.

Plantation forestry is not likely to allow the retention of much of the original biodiversity, but may provide an important buffer area of little-used land and may help to maintain the populations of wide-ranging species with the largest area requirements (e.g. elephants, tigers). There may be several ways in which the biodiversity value of plantations could be enhanced: (i) by changing plantations from monocultures into polycultures; (ii) by creating mosaics of plantations of different species or species combinations; and (iii) by integrating them with patches of natural forest which are to be maintained on steep slopes, ridge tops and along waterways. As of yet, little research is available on this topic.

ONE POSSIBLE SOLUTION: INTEGRATED LAND USE

The previous section concluded that the land area currently allocated to protection is unlikely to help in maintaining future biodiversity levels. However, we also saw that some economic activities may not radically modify an area's natural vegetation and only cause small reductions in biodiversity. Where such forms of land-use are adjacent to protected areas, the ensuing complex may have greatly enhanced conservation potential. For example, a timber plantation adjacent to a major reserve is bound to have a much greater value for biodiversity preservation than one surrounded by agricultural land. Not only will it accommodate certain species that require large ranges and may be at risk of local extinction without the reserve, but immigration from the reserve may also prevent the local populations in the plantation from going extinct.

Spatial integration of biodiversity-compatible forms of land-use is therefore an essential planning instrument for long-term preservation of biodiversity. The most effective application of this concept is in *integrated land-use complexes*. These complexes consist of a central core area surrounded by economic activity zones of varying use intensity. As one moves from the core to the periphery of these complexes, the policy priorities shift from preservation of species (strict biodiversity protection) to sustainable production of goods and services (broad biodiversity protection).

Ecological Considerations in Designing Integrated Land Use Complexes

Components of Integrated Land Use Complexes

The central component of a complex is a *core reserve area*. In this analysis, the core protection area is equivalent to the "reserve" discussed extensively in Chapter 3. Most of the parameters discussed in that context apply as well to designing core reserve areas.

Surrounding the core reserve(s) are *economic activity zones*. The concept of economic activity zones is an extension of the idea of buffer zones. Buffer zones were initially proposed by the UN's Man and Biosphere program as a part of their Biosphere Reserves (Gregg and McGean 1985). These buffer zones arose in response to the encroachment of rural poor on major conservation areas. Implementation of buffer zones is still in the early stages.

Economic activity zones are an expanded form of buffer zones that not only protect the core area, but also provide economic returns to local peoples (MacKinnon et al, 1986). These expanded zones around the core conservation area provide for sustainable exploitation of important resources for the local population, and support low-impact activities that help to increase population numbers of threatened species and the overall biodiversity of the complex.

A good example of policy concerning these expanded use zones can be seen in Indonesia, where buffer zone development is turning into a more encompassing Integrated Conservation Development Program (Wind 1991). This program involves bottom-up planning, the development of local planning institutions, and community participation, in addition to the development of a number of sustainable harvesting and development activities (reforestation and rehabilitation of damaged ecosystems; forest plantations; natural forest management; extractive use for non-timber forest products; smallholder estates; cash tree crop plantations; transmigration and resettlement programs; village development programs).

Box 4-1:
Extended Complexes

The following ingredients may be necessary for successful implementation of these complexes. A complex requires a natural core covered with pristine or mainly pristine tropical rain forest, which is:

- of appreciable size, 500,000 ha or more, if possible;
- located in a region with high biological diversity and/or high local endemism;
- composed of a complete array of natural habitats, where possible covering major altitudinal, soil and rainfall gradients;
- surrounded by an area with predominantly natural vegetation, and covering at least 1 million ha;
- low in human population density so as to prevent acute conflicts over land and to sustain the kinds of land use that preserve biodiversity (see below).

Integrating the Components in a Complex

The relative size of core areas versus economic activity zones and the appropriate activities in these zones will strongly depend on human population density in the region. Where tropical rain forests are still the dominant feature on the landscape, the priority for integrated land-use complexes can be biodiversity preservation. In contrast, where tropical rain forests have become "islands" in an area dominated by agriculture, the emphasis should primarily be on maintaining essential ecological processes and only secondarily on strict biodiversity preservation.

Sustainable extraction of timber and non-timber forest products can be undertaken on a very large scale only in thinly populated regions (Salafsky et al. in press). In these areas of low population density, special opportunities for biodiversity preservation exist. For instance, it may be possible to combine spatially individual core areas and their associated economic activity zones into larger complexes, especially along watershed lines. These *extended complexes* are likely to maintain their biodiversity in perpetuity and be fairly resilient to the effects of global climate change. Each biogeographic province ought to have at least one such extended complex. In Box 4-1, we list some of the considerations that should go into establishing these extended complexes.

At intermediate population densities, land uses such as ecotourism or natural forest management for timber may be more appropriate, as they provide substantial economic returns while also preserving biodiversity. In areas of high population density, however, strict biodiversity protection should be secondary to maintaining essential ecological processes. Nonetheless, spatial integration of land-use according to zoning principles is valid in every kind of landscape containing natural elements (McNeely et al. 1990). A dense human population, poverty, inefficient agriculture and high rates of encroachment into areas that are unable to sustain permanent agriculture all go hand in hand. Agroforestry may help to reduce the erosion of biodiversity in such areas, but solutions may have to be found outside conservation areas. Ultimately, in these densely populated regions, only intensification of agriculture, industrial development, and family planning can relieve the pressures on natural areas.

Within the framework of integrated land-use complexes, damaged or regenerating eco-systems may provide unsuspected potential for indirect biodiversity preservation. Their presence is not necessarily restricted to areas with high human population density. For instance, large areas in many tropical rain forest countries are covered by coarse grasses and shrubs. These scrublands are maintained by fires set by pastoralists to provide new shoots that are palatable to grazing animals. In areas of low population density, such degraded ecosystems could potentially be rehabilitated by encouraging ecological succession

toward secondary forest. Biodiversity of secondary forests is much lower than that of pristine forests (Terborgh and Weske 1969), and to actually restore biodiversity in areas after large-scale disturbance may take centuries (Wilson 1988); however, secondary forests do contain many elements of the original forests, and thus contribute to increasing population sizes of many species. Also, secondary forests often harbor high densities of large browsing herbivores, which can serve as important sources of meat for local communities.

The inclusion of regenerating forests may be of great value to the land-use complex in another important way. The critical minimum size of the complex is set by species with the largest area requirements. These animals are often the ones that are least specific in their habitat requirements (Harris 1984). Examples include the big browsers and the top carnivores (the possible exception being raptors) which tend to do relatively well in secondary forests. Although the species that limit the minimum size of the complex will vary from place to place, and need not always be the ones that thrive in secondary forests, the potential value of connecting secondary forests to the core may be great.

In summary, integrated land-use planning can greatly improve the protection of the reserve core, as well as enhance the biodiversity value of the complex as a whole. If the peripheral areas contain important natural elements, they increase the population sizes of at least some of the species with critical area needs, and may also increase the total number of species contained in the complex (Harris 1984; McNeely et al. 1990). The extent to which integrated land-use planning contributes to preserving biodiversity in addition to protecting existing reserves largely depends on the existing human impact on the landscape. This contribution will be greatest where tropical rain forests still form the matrix of the landscape instead of being islands in it. Where forests have become islands, integrated land-use planning mainly will help to safeguard these islands from over-exploitation and the ensuing loss of ecological goods and services (as well as biodiversity).

Economic Considerations in the Development of Integrated Land Use Complexes

The primary criterion for deciding where to establish integrated land-use complexes undoubtedly will be the standing biodiversity of the region. Economic considerations, however, constrain both the total amount of land that can be preserved and the location of the preservation sites. A number of decision making criteria can be used to decide which areas to protect.

Safe Minimum Standards

One way to incorporate economic factors into ecological concerns about irreversibilities is the *safe minimum standards* (SMS) approach discussed in section 2.1. According to this criterion, protection of a region which is biologically unique should be undertaken so long as the social costs of protection are not unacceptably high (Bishop 1978). In general, proponents of SMS argue that economics cannot by itself determine what are "unacceptably high costs". However, once the choice of the region is made, economists will be needed to evaluate the net costs (costs minus benefits) of protection and establish the extent and implications of these costs. In the context of planning integrated land use complexes, under an SMS approach, economics comes into play once an area has been chosen based on ecological considerations (high diversity, high endemism etc). After this choice has been made, the size of the protected core can be established by determining the optimal size given economic and ecological constraints.

Benefit Cost Analysis

Benefit Cost Analysis (BCA) is another important economic criterion which can be applied to biodiversity protection. Benefit cost analysis would imply a consideration of all the benefits and costs of protection in different potential regions for protection. For example, if a number of different reserves or national parks are planned, the benefits and costs allocatable to the different reserves would need to be established. The *Net Present Value* (NPV) of benefits (the discounted value of benefits minus costs) accruing from protecting each of the different reserves would have to be considered, and the protected area with the highest Net Present Value selected. This results in economic efficiency. A sound argument for using BCA for making policy decisions can be made when all the benefits and costs are known and estimable (Randall 1991). However, a complete accounting of the benefits of strict biodiversity protection is probably an impossible task.

Using SMS to approach the biodiversity protection problem does not by any means preclude the use of BCA. As mentioned earlier, BCA can be a criterion for decision making where one has to choose between two or more reserves of comparable biodiversity.

BCA can play another important role in examining distribution of benefits and costs. This *distributional analysis* charts the flow of resources across groups which benefit or must bear the costs of a policy decision. This can be useful and even critical to the success of integrated land use programs. In the next section we present a detailed analysis of the distribution of costs and benefits for integrated land use.

Cost Effectiveness

A third economic decision making criterion applicable to the issue of biodiversity protection is cost effectiveness. Conducting a benefit cost analysis for different potential reserves may not always be feasible, in which case, cost effectiveness analysis can be undertaken. Cost effectiveness implies an estimation of the costs of protection in different potential reserves, and the selection of the least cost option. It is thus a variant of BCA that focuses only on the cost side of the equation. Cost effectiveness can serve as a criterion for choice of protected areas, but it is better established as a technique for assessing different management plans for protection (see Box 4-2 for an example of the potential use of cost effectiveness for deciding between different management options).

Summarizing the implications of incorporating economic considerations into integrated land use planning, biodiversity protection should be undertaken such that:

1. A region of high biodiversity is chosen based on a relative ranking of areas which have high diversity, high endemism etc. If biodiversity is to be protected, then regions with maximum biodiversity have to be given maximum priority.

2. Net economic costs are treated as a constraint to protection. Therefore the optimal land size for the core and for buffer zones is constrained by economic considerations. If biodiversity protection is to be successful, then economic considerations need to have direct relevance in establishing land-use zones and corridors.

3. If there are several regions or areas which are equally ranked on a biodiversity scale, then the region which maximizes net economic benefits is chosen. Thus, to protect strict biodiversity, economic consideration will play a secondary role unless a choice between two or more equally diverse areas has to be made.

Box 4-2: Cost Effectiveness and the Preservation of Biodiversity

Hyman and Wernsted (1991) use cost effectiveness analysis to address the choice between different management plans for preservation by the administrators of the Endangered Species Act. They applied a biological model combined with an economic model to evaluate the cost effectiveness of various management actions aimed at mitigating the damage done to endangered salmon populations in the Snake river. With these kinds of mathematical models, a cost effective frontier can be developed. This frontier maps different management plans with reference to costs and effectiveness (probability of success of the plan or the probability of the survival of different species). All plans which lie beneath the frontier are discarded, and a plan on the frontier is chosen based on some pre-determined priorities. Use of cost effective techniques results in economic efficiency when circumstances are such that there is a clear indication that the benefits of preserving biodiversity are high.

Distributional Considerations in the Development of Integrated Land-Use Complexes

So far we have established that integrated land-use complexes could be a ecologically viable means for "strict" biodiversity protection. Furthermore, economic considerations need to be treated as critical constraints in planning these complexes. In this section we examine the distribution of benefits and costs from the establishment of integrated land-use complexes, distinguishing between those benefits and costs accruing from core reserve areas and those from surrounding buffer zones.

What Are the Benefits of Protection?

Earlier, we outlined two different types of benefits from strict biodiversity conservation in core reserve areas: risk minimization, and existence values. In considering integrated land-use complexes, additional broad biodiversity benefits are generated as well as indirect benefits, as buffer zones protect biodiversity in the core area. As outlined in Table 1, these benefits can potentially accrue to three groups: local peoples, regional and national residents, and the international community.

Local peoples generally will derive benefits from the establishment of the core through reduced risk of the entire complex crashing with its services and its products, some revenues from ecotourism and recreation, and existence values (assuming these are positive). There may also be some small benefits in terms of infrastructure-related employment. Local peoples will generally derive large benefits, however, from development of buffer zones (unless establishment of these zones actually limits their use of a traditional harvesting area).

Regional and national residents will generally benefit from core reserve areas. These benefits include both revenues from and opportunities to participate in ecotourism and scientific research, benefits from reduced risk in loss of watershed protection and other ecological services, and any existence values they may have. These residents may also obtain some benefits from enhanced productivity in buffer zones in the form of products for consumption and enhanced national income (if buffer zones are managed such that there are net productivity gains).

Finally, *residents of the global community* will receive extensive benefits from the core reserve area in the form of tourism, scientific opportunities, and existence values for biodiversity. This group, however, only indirectly benefits from the establishment of buffer zones (to the extent the buffer zones are critical for the continued existence of the core areas).

What Are the Costs of Protection?

Costs associated with the formation of integrated land-use complexes can also be seen as accruing to local, regional or national, and international groups (Table 1).

Local people will generally bear most of the costs of establishing reduced access core areas. These opportunity costs will reflect the degree to which they currently use the lands incorporated into the core area. Furthermore, as mentioned above, they will also bear the cost of buffer zone creation to the extent that it limits previously existing land-uses and prevents them from expanding their farming activities. Local peoples may also incur costs of crop damage from animals in the protected areas (which can either be actual crop losses, or time and money expended to protect crops). Finally, local peoples may suffer from negative effects of tourism in the region, including erosion of traditional cultures or increased competition within existing markets.

Regional and national residents will as a group incur the costs of development of required infrastructure in an integrated land-use complex. These costs include paying salaries for park personnel, building tourist facilities, funding resource management agencies, and providing economic incentives to locals for using the buffer zones. They may also bear the (presumably short-term) loss of timber and other product revenues from the national income account.

By contrast, the *international community* will not bear any of the costs involved in the development of integrated land-use complexes unless specific arrangements are made to share in the costs outlined above. If the extent of the dependence of the core on the buffer zones can be established, then the economic impacts of core protection would include a percentage of the costs from the development of the buffer zones. These costs along with the direct costs of protecting the core reserve area could potentially be borne by the international community.

Who Should Pay? The Financial Implications of Strict Biodiversity Protection and Mechanisms for Payment

Overall, the above analysis indicates that with the exception of risk reduction benefits, benefits from establishing integrated land-use complexes accrue primarily to the global community and secondarily to regional and national residents. Costs, on the other hand, accrue primarily to local residents and secondarily to regional and national residents. Accordingly, if conservation is going to occur, recipients of benefits must compensate those who bear the direct costs of strict biodiversity protection in core areas and associated buffer zones. In other words, ***strict biodiversity conservation will only occur if the global community is willing to pay local, regional, and national peoples to preserve it (see Box 4-3).***

In particular, analytical tools are needed that estimate the additional costs of protecting strict biodiversity over and above the costs of safeguarding the ecological services and other products (broad biodiversity). (As the latter investments produce fairly immediate economic benefits, they are likely to be made anyway.) A variety of methods have been proposed to generate funds for biodiversity protection. Among these are increased user fees for parks and reserves, special taxes (e.g. water charges for downstream users, taxes on hunting equipment), and voluntary contributions from nature tourism and other resource based industries (McNeely et al. 1990). However, these sources are unlikely to generate sufficient funds for large-scale strict biodiversity preservation projects. Moreover, the taxes do not affect the primary beneficiaries of strict biodiversity protection since they generally live far beyond the boundaries of a given park, province, or country.

TABLE 1
Integrated Land Use Planning- The Distribution of Costs and Benefits

Item	LOCAL				NATIONAL				INTERNATIONAL			
	Core		Buffer Zones		Core		Buffer Zones		Core		Buffer Zones	
	B	C	B	C	B	C	B	C	B	C	B	C
1.Existence Values	+				++		i		+++		i	
2.Risk Reduction	++				++		+		?		i	
3.Nature Tourism & Scientific Opportunity	+	-			++	--			+++	-	i	
4.Forest Use		---	+++	--		-		-				
5.National Income						-	+	?				
6.Infrastructure & Management	+					---		--		???		?

B=Benefits C=Costs + indicates relative magnitude of benefits - indicates relative magnitude of costs i indicates indirect benefits

? indicates ambiguous benefits or costs

International cooperation is needed to overcome financial constraints in the countries where tropical forests are located, and to assess beneficiaries worldwide for contributions to biodiversity protection efforts. Three methods of international cooperation are discussed below: Debt-for-Nature Swaps, the recently established Global Environmental Facility, and a proposed Global Nature Fund.

Debt for Nature Swaps

Debt for nature swaps are arrangements under which a conservation organization purchases a developing country's debt at a discounted value in a secondary debt market (Hansen 1988). The purchased debt is then canceled in exchange for the debtor country undertaking an environmentally related activity. Often, the agreement is for the country to take local currency in the amount of the debt's face value and invest it in conservation activities managed by local organizations. Since a number of less developed countries have both large debts and significant amounts of tropical forests, organizations such as the World Wildlife Fund and Conservation International (CI) have taken this opportunity to facilitate habitat preservation at a relatively low cost. However, the arrangements have been criticized as foreign impingement on debtor nations' sovereignty. CI arranged the first debt for nature swap in Bolivia in 1987 (McNeely 1988). Since then, swaps have occurred in Madagascar, Ecuador, Zambia, and other countries.

Global Environment Facility (GEF)

Implemented jointly by the United Nations Development Program (UNDP), the United Nations Environment Program (UNEP), and the World Bank, the GEF is an effort to provide international cooperation and financing for large scale environmental projects in developing countries. With initial three year funding of $1.5 billion, the GEF was established in 1990 with four major program areas: reducing global warming, preserving biological diversity, protecting international waters, and slowing depletion of the ozone layer. Approximately 20 countries have contributed to the fund. The World Bank administers the fund and is responsible for appraising and supervising projects. Among the approved projects are biodiversity projects in East Africa and the South Pacific, and a tropical forest protection project in the Congo.

Global Nature Fund

Given the array of demands on the GEF and its rather limited funding, it may be desirable to establish a separate or parallel fund, called the Global Nature Fund (GNF), with the specific purpose of supporting forest conservation and sustainable management (Sharma and Kramer 1990). The GNF would be established to supplement the international funding for forestry activities available through the Tropical Forestry Action Plan and other current sources.

The GNF could be supported by several alternative funding mechanisms. A funding commitment from all countries with at least US$6,000 of per capita GNP could be requested. An assessment of one-tenth of one percent of GNP would generate about US$11 billion and could be replenished every three years. Another funding approach would be an import tax in the industrial countries on tropical timber products. The fund could be used for investments supporting (i) forest conservation (reserves, parks, etc.), (ii) sustainable management of natural forests, (iii) reforestation and afforestation activities, and (iv) institution building and human resource development. All of these activities could contribute to biodiversity protection, if such protection were made a specific goal of the program.

Box 4-3:
Is Biodiversity a Luxury Good?

A luxury good is a good which is highly income elastic. That is, as incomes rise the demand for such goods increases more than proportionately. We would argue that strict biodiversity is one such good. The demand for strict biodiversity has two independent components to it - risk reduction value and existence value. The value of risk reduction emerges from the interlinkages between various components and functions of nature which are supported by strict biodiversity, and which in turn support the flow of ecological services. The continued existence of the ecosystem in consideration depends on these interlinkages and the flow of services within the system. If we assume that people are on average risk neutral or risk averse, a decline in risk implies increased benefits. There is a considerable amount of literature which suggests that at low income levels, people are risk averse. This then suggests that the returns from reduced risk may be considerably important at low income levels. To see this let us consider the case of a poor subsistence farmer living within an ecosystem which has the potential to change from tropical rain forests to derived savannah. Now suppose that she is offered two lotteries. In the first, forests are allowed to degrade but she knows there is a 50% chance of returns to agriculture being worth $50 because the local government is distributing fertilizers and a 50% chance of a decline in food production by $40 because the local superintendent captures the money. The second lottery has a certain return of $ 5 (i.e no loss in agricultural returns because the ecosystem is healthy). The poor farmer is more likely than her wealthy neighbor to pick the certain return over the uncertain lottery, even though both have the same expected return.

Now consider the other component of the demand for strict biodiversity, existence, or non-use, values. In as much as non-use values are income constrained, they are most likely normal goods (i.e. the willingness to pay for these goods increases with increases in income). A poor farmer may enjoy the variety of animals in the forests around her village, but if given the choice between forgoing one extra meal and preserving 50 extra animals her decision is very clear. However, for her wealthy neighbor the same decision is not as clear. Non-use values are normal but are they income elastic? Does willingness to pay increase more than proportionately with income? There are a number studies in the developed world which estimate non-use values for different environmental goods which suggests that they are in general income elastic.

In sum, we conclude that with increases in wealth and income the demand for biodiversity is likely to increase more than proportionately. It can be argued that the reduced risk benefits may dominate the other benefits. However, in as much as these benefits are likely to be realized in the future, they will need to be discounted and this in itself will result in decreasing the value of risk reduction.

What then are the implications of strict biodiversity being a luxury good for biodiversity conservation? One important implication stands out: by conserving biodiversity we are disproportionately benefitting the rich relative to the poor.

All developing countries would be eligible for funding. To apply for GNF funding, countries would need to demonstrate strong commitment to sustainable management of critical forest ecosystems. The fund could be managed by the World Bank and the regional development banks (Asian Development Bank, African Development Bank, and Inter-American Bank), with active involvement of NGOs. Through international cooperation, the Fund would combine financial commitments from the world community with resolve from sovereign nations to use and manage forests more efficiently for present and future generations.

Implementation of Integrated Land Use Complexes

Many different issues must be addressed in implementing integrated land-use complexes. First, the ecological infrastructure must be studied. The starting point should be a protected area of outstanding biodiversity. Usually, one or more existing reserves will form the basis for the core area. It must be established whether this core is adequate in terms of size and variability of topography, habitats and altitude. If there is considerable pressure on the park or its surroundings, the leading causes of current deforestation must be identified in the region. If, for whatever reason, the underlying processes can not be stopped or redirected, any attempt to develop integrated land-use will be futile.

Next, existing forms of land use need to be mapped in the whole region. If local people use the core area, this use must be detailed and alternative uses should be sought elsewhere, on the basis of input from the local population. Land ownership and possible conflicts between local people and central governments on this issue must be clarified (Van Orsdol 1987). Information on population growth (spontaneous and through immigration) is needed to assess future pressure on the landscape and to predict where bottlenecks are likely to arise. Existing and expected damage by wildlife to crops should be estimated.

In the next phase, the potential of each part for ecotourism and the selective extraction of timber and non-timber forest products (for direct consumption and for trade) must be determined. Particular attention should be given to the feasibility of creating corridors of utilized forest between existing forest blocks. Where relevant, the role of indigenous people needs to be examined. Existing plans must be reviewed for their compatibility with integrated land-use planning. Links with existing programs should be explored (i.e. the national versions of the Tropical Forestry Action Plan, or national and international reforestation initiatives). Designated land use must be clear for each area, as comparisons of planning maps from different agencies or different levels of government often reveal incompatibilities.

One organizational point of crucial importance is the control of "illegal" land use. Many tropical nations already have central planning agencies that make detailed land-use plans. The integration of different forms of land use for biodiversity preservation is a natural part of their mission, and they need ecologists on their staff to implement this task. However, the greatest need is in the field. Historically, in virtually all developing nations with extensive tropical forests, there has been a massive transfer of land control to the central government, a trend that is not always recognized by the local population (Poffenberger 1990). The conflict over land tenure and title between local people and the government makes conflict over land use almost inevitable. These conflicts must, in one way or another, be resolved if any planning of land use is to work at all. Adequate enforcement in the remote areas is only possible with the cooperation of the local population, and therefore some form of local involvement in the planning stages is crucial for the success of the enforcement of land-use plans.

Finally, as with all development programs, periodic evaluation and monitoring of the extent to which program goals are being met is needed. Again, continuous feedback from the local communities is essential.

CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER ACTION

This paper draws two major conclusions. First, ecologically and economically sustainable development will by itself not stop the erosion of tropical rain forest biodiversity caused by fragmentation, exploitation, and human activities. Therefore, strict biodiversity preserving action is required and should be spatially integrated with other forms of land-use which have a limited adverse impact on biodiversity. Second, while the costs of biodiversity preservation accrue mainly at the local level, the benefits accrue mainly at the regional, national, and especially global level. Therefore, special mechanisms need to be devised to fund this biodiversity preservation.

Both these conclusions contain elements of uncertainty. No ecologist knows exactly how many rain forest species will go extinct under current conditions. Likewise, no economist knows the full value of the benefits of biodiversity protection. Nonetheless, most experts agree that we cannot wait until this information is available. Instead, a more pressing need is to use existing ecological and economic theory to develop and evaluate practical techniques of biodiversity preservation such as integrated land-use complexes. The following is a list of priorities for applied research and action.

1. Forest Exploitation and Biodiversity Protection

Problem:
For most forms of exploitation of tropical rain forest, the extent to which they are compatible with strict biodiversity preservation is unknown.

Suggested Research or Action:

a) Evaluate the effects on biodiversity of existing techniques of using natural forest for extraction of timber and non-timber forest products and for ecotourism;

b) Explore ways to improve these techniques to enhance their compatibility with biodiversity preservation within economic constraints. Examples of relevant questions are: What are the effects on biodiversity of leaving pristine forest islands? Can zoned clear-cutting be made compatible with biodiversity preservation?

c) Assess the impact on biodiversity of tree-oriented land-use systems such as agroforestry areas, timber plantations, tree-crop plantations, and secondary forest.

2. Benefits, Costs, and Distributional Impacts of Biodiversity Protection

Problem:
The magnitude and incidence of the benefits of biodiversity protection from the establishment of integrated land-use complexes are currently unknown. Furthermore, the opportunity costs of establishing such complexes are unknown.

Suggested Research and Action:

a) Estimate the benefits and costs at different levels (local, regional and national, and international) of establishing these complexes (see Table 1);

b) Devise mechanisms to transfer benefits from the beneficiaries to those who bear the costs;

c) Develop alternative policies or land-use activities that can be undertaken in high biodiversity areas to create incentives for biodiversity preservation;

d) Determine to what extent governments are responsible for the high opportunity costs involved in biodiversity preservation through the effects of taxes, subsidies, and price distortions, and create policies that rectify the situation.

3. Implementation of Integrated Land-Use

Problem:
How are all these aspects to be combined into workable policies such as in designing and administering integrated land-use complexes?

Suggested Action and Research:
a) Establish the physical and economic *(extent of the market)* boundaries for integrated land-use complexes in areas with varying ecological and socioeconomic conditions;

b) Develop pilot projects in such areas, combining core protected areas with economic activity zones. Once the necessary ecological and economic research is conducted, design practical land-use plans that are both ecologically and economically sound;

c) Identify and develop mechanisms by which projects for protecting biodiversity involve the active participation of all the stakeholders, i.e. local people, NGOs, and government agencies;

d) Identify areas of conflict between ecological and economic goals, and between different stakeholders. Determine techniques and options for resolving these conflicts.

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