BIODIVERSITY AS AN ENVIRONMENTAL SERVICE IN BRAZIL'S AMAZONIAN FORESTS: RISKS, VALUE AND CONSERVATION

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SUMMARY

The environmental service provided by Amazonian forests in maintaining their great wealth of biodiversity is one of several factors leading to the conclusion that much greater efforts are warranted to reduce the destruction of these forests. Risks to biodiversity in Amazonian forests include deforestation, logging, fires, fragmentation, depletion of fauna, invasion by exotic species, and climate change. Financial values assigned to biodiversity depend strongly on the purposes of valuation. Utilitarian benefits include the values of presently-marketed and presently-unexploited forest products, and the monetary value of environmental benefits. Non-monetary values of Amazonian forests are also essential components of decision-making on conservation. Measures of "willingness to pay" and "willingness to accept" can be useful as indicators of potential financial flows, but should not be confused with the true values of the forests to society. Valuation for the purpose of setting penalties for destruction of biodiversity is an important legal question in Brazil and must take into consideration additional factors.

Conservation of biodiversity in Brazil includes creation of various types of protected areas. The status of these areas varies greatly, with practice frequently deviating from official requirements. Creating reserves that include human occupants has a variety of pros and cons. Although the effect of humans is not always benign, much larger areas can be brought under protection regimes if human occupants are included. Additional considerations apply to buffer zones around protected areas. The choice and design of reserves depends on the financial costs and biodiversity benefits of different strategies. In Brazil, rapid creation of lightly protected "paper parks" has been a means of keeping ahead of the advance of barriers to establishment of new conservation units, but emphasis must eventually shift to better protection of existing reserves. Indigenous peoples have the best record of maintaining forest, but negotiation with these peoples is essential in order to ensure maintenance of the large areas of forest they inhabit. The benefits of environmental services provided by the forest must accrue to those who maintain these forests. Development of mechanisms to capture the value of these services will be a key factor affecting the long-term prospects of Amazonian forests. However, many effective measures to discourage deforestation could be taken immediately through government action, including levying and collecting taxes that discourage land speculation, changing land tenure establishment procedures so as not to reward deforestation, revoking remaining incentives, restricting road building and improvement, strengthening requirements for environmental impact statements (RIMAs) for proposed development projects, and creating employment alternatives.

Running Head: Biodiversity in Amazonia

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BIODIVERSITY AND AMAZONIA

"Biological diversity," or "biodiversity," refers to the total variability of life on Earth (Heywood & Watson 1995, p. 5). It includes variation not only at species level, but also at other taxonomic levels and at the genetic level (for example within single populations), as well as variation in ecological functions such as those of pollinators and seed dispersers. The Convention on Biological Diversity, opened for signature at the United Nations Conference on Environment and Development (UNCED or ECO-92) in 1992, came into force in December 1993 with the objective of furthering global conservation of biodiversity and its contribution to sustainable development (but see Guruswamy 1999). The Global Biodiversity Assessment, commissioned by the United Nations Environment Programme (UNEP) in 1993 and completed in 1995, provides baseline scientific information for implementation of the Convention (Heywood & Watson 1995).

Brazil is one of the five "megadiversity" countries in the world recognized by the World Wide Fund for Nature (WWF) (Mittermeier 1988). According to the 1990 Forest Resources Assessment by the United Nations Food and Agriculture Organization (FAO), Brazil has 41% of all remaining forests classified as tropical rain forest (FAO 1993). Brazil as a whole has an estimated 55,000 angiosperm plant species, more than any other country (McNeely et al. 1990). Brazil as a whole has 428 species of mammals, placing it third in the world. Brazil has 1622 bird species, a number exceeded only by Colombia and Peru, while Brazil's 516 species of amphibians is the world's greatest number in a single country (McNeely et al. 1990). Similarly, Brazil's butterflies and reptiles place the country in 4th place with 467 and 74 species, respectively (McNeely et al. 1990).

Invertebrates make up by far the largest share of total biodiversity. The canopy of trees fumigated with malathion in four forest types near Manaus yielded 1080 species of beetles (Coleoptera) in 61 families, with only 1% overlap in species found at sites only 70 km apart (Erwin 1983, 1988). Together with similar studies made on a larger scale in Peru and Panama, these collections have more than tripled the total number of species of organisms estimated to exist on Earth (Erwin 1982, 1988). With extrapolations from single trees to the globe, however, the sample sizes are so small that little confidence can be attached to numbers. The fact that the arthropod fauna is tremendously diverse is incontestable, however.

RISKS TO BIODIVERSITY IN AMAZONIAN FORESTS

Deforestation

LANDSAT satellite data interpreted at Brazil's National Institute for Space Research (INPE) indicate that by 1998 the area of forest cleared in Brazilian Amazonia had reached 547,200 km^2 (13.7% of the 4 million km^2 originally forested portion of Brazil's 5 million km^2 Legal Amazon region), including approximately 100,000 km^2 of "old" (pre-1970) deforestation in Pará and

Maranhão (Brazil, INPE 1999). Brazil's Legal Amazon region (Fig. 1) is about the size of Western Europe, and the area that has been deforested so far is the size of France. The rate of forest loss (including hydroelectric flooding) declined from 1987 to 1991, after which it rose again, made a sharp jump in 1995 followed by a decline through 1997 and rose again in 1998 (Fig. 2).

[Figures 1 and 2 here]

Interpretation of the causes of deforestation suggested by Brazilian deforestation data strongly influences any conclusions that may be drawn regarding whether it is feasible to reduce deforestation and what countermeasures might be most effective. Recently, the head of the Brazilian Institute for the Environment and Renewable Natural Resources (IBAMA) interpreted these data as indicating that deforestation is now primarily the work of landless peasants and small farmers (Traumann 1998). INPE has interpreted the numbers for 1997 and 1998 in the same way (Anonymous 1999a). Were this the case, substantial reductions in clearing rates would not be possible, or would be difficult and expensive, without exacerbating poverty in the region. However, four independent lines of evidence indicate that it is still the rich, rather than the poor, who are responsible for the bulk of Brazil's deforestation. One indication is the close correspondence of the major swings in deforestation rates with macroeconomic changes that affect investors rather than small farmers using family labour. The decline in deforestation rates from 1987 through 1991 can best be explained by Brazil's deepening economic recession over this period; ranchers simply did not have money to invest in expanding their clearings as quickly as they had in the past. At the low point in 1991, investors were still without access to much of their funds because then-President Fernando Collor had frozen bank accounts in the country in 1990. The peak in 1995 is best understood as a reflection of economic recovery under the Plano Real, a set of economic reforms implanted in July 1994 that resulted in larger volumes of money suddenly becoming available for investment, including investment in cattle ranches. The fall in deforestation rates in the years after 1995 is a logical consequence of the Plano Real having sharply cut the rate of inflation. Land values reached a peak in 1995, and fell by about 50% by the end of 1997. Falling land values made land speculation unattractive to investors. An additional factor reducing the value of land was a substantial increase in the rural land tax (ITR) for areas classed as “unproductive”.

The second line of evidence that medium and large ranchers are the major deforestation agents is the distribution of clearing activity among the region's nine states; this indicates that most of the clearing is in states that are dominated by ranchers: the state of Mato Grosso alone accounted for 26% of the 11,100 km^2 total in 1991 (Brazil, INPE 1999). Mato Grosso has the highest percentage (84% at the time of the last [1985] agricultural census) of its privately held land in ranches of 1000 ha or more. By contrast, Rondônia, which is a state that has become famous for its deforestation by small farmers (Malingreau & Tucker 1988; Skole & Tucker 1993; Skole et al. 1994)--had only 10% of the 1991 deforestation total, and Acre had 3% (Brazil, INPE 1999). The number of properties censused in each size class explained 74% of the variation in deforestation rate among the nine Amazonian states in both 1990 and 1991

(Fearnside 1993). Multiple regressions indicate that 30% of the clearing in these years can be attributed to small farmers (properties <100 ha in area), and the remaining 70% to either medium or large ranchers (Fearnside 1993).

The third line of evidence is data released by INPE (Brazil, INPE 1998, 1999) indicating that only 21% of the area of new clearings in 1995, 18% in 1996 and 10% in 1997 were <15 ha in area. Note that these values refer to the areas of new clearings, as distinct from the areas of the properties in which the clearings are located. Small farmer families are only capable of clearing about 3 ha yr^{-1} using family labour (Fearnside 1980a), and this is reflected in deforestation behaviour in settlement areas (Fearnside 1984).

The fourth line of evidence is direct observations and interviews with farmers and ranchers. A study of 202 properties distributed among different size classes and among five sub-regions in Brazil's ‘arc of deforestation’ that extends from Paragominas (Pará) to Rio Branco (Acre) concluded that in the 1994-95 period only about 25% of the clearing activity was in properties of 100 ha or less (Nepstad et al. 1999). Together, these lines of evidence indicate that it is a myth that the bulk of Brazil's Amazonian deforestation is carried out by people who are clearing to feed themselves. The predominance of medium and large ranchers in Brazilian Amazonia means that deforestation could be substantially reduced without worsening the plight of the poor.

Deforestation has a severe impact on biodiversity in tropical forest areas because most of the species present are unable to survive the radical changes brought about by cutting and burning the forest. Because many tropical forest species are endemic to restricted areas, the clearing of extensive tracts of these forests can be expected to lead to extinction of these species. The risk of extinction of any given species is not a linear function of the remaining area of habitat, but rather rises sharply as the remaining area approaches zero. Relatively small remnants, such as the remaining vestiges of Atlantic forest, can harbor a surprisingly large share of the original suite of species (Brown & Brown 1992; Primack 1998, p. 168). However, once the forest reaches this critical state, each hectare of additional forest loss has severe impact on species survival (Heywood & Stuart 1992).

A relationship between species numbers and area has long been recognized (MacArthur & Wilson 1967). Probabilities of extinction can be assessed quantitatively from time-series data using population viability analysis (PVA), but results are often misleadingly optimistic if data series are inadequate or if analyses do not include the very important role of catastrophes (Ludwig 1999). As populations dwindle, the chance of random events driving populations to extinction increases dramatically (e.g. May 1973).

Logging

Logging has severe direct impacts on the forest. Mahogany (Swietenia spp.) is particularly vulnerable to local extinction because of its population structure, which consists of concentrations of even-aged adult trees with no stock of seedlings or young trees. Because of its great economic value, mahogany logging plays a catalytic role in initiating a process of destruction that leads to loss of the entire ecosystem (Fearnside 1997b).

Logging has indirect impacts through access road construction and by providing funds to ranchers for expansion of clearings. It also makes the remaining forest much more susceptible to fire by accidentally killing many unharvested trees, by opening the forest canopy and by leaving large quantities of dead wood in the forest. This has been documented in logging areas in Eastern Amazonia (Uhl & Buschbacher 1985; Uhl & Kauffman 1990; Nepstad et al. 1999). Logging has been rapidly expanding in many parts of the region, and can be expected to increase even more rapidly in the future as Asian forests are no longer able to supply the volume of wood demanded by global timber markets.

Natural forest management for timber has often been promoted as a means of maintaining biodiversity. However, the effects of the logging in these systems can have severe consequences for biodiversity, suggesting that other strategies for maintaining forests should be pursued (Bawa & Seidler 1998). So far, logging impacts have fallen short of causing extinctions (Johns 1997).

Fires

The Great Roraima Fire from September 1997 to March 1998 brought attention to the importance of climate as a factor to be reckoned with in efforts to maintain Amazonian forest. Entry of fire into forest greatly increases the risk of much more severe fires entering the same forests in future years, initiating a positive feedback process that can destroy the forest completely (Cochrane & Schulze 1999). El Niño events, such as that of 1997/98, are natural oscillations not caused by human activity which, when added to the effects of human action, can lead to burning of standing forest (Cochrane et al. 1999). Most of the forest that burned in Roraima was in areas that had not experienced any significant amount of logging activity (Barbosa 1998); this should serve as a warning of the danger present in the rest of the region, where the conditions for starting fires in standing forest can be reached with greater frequency.

Roads and settlements adjacent to forested areas that are intended to be protected (such as the Yanomami Indigenous Area) provide sources of fire that have a high probability of escaping control and entering the forest. Roads and settlements are built by government authorities, who make decisions to implement them without taking into account the impacts of fire beyond what can be expected to be deliberately cleared and burned for agriculture (Barbosa & Fearnside nd). Decisions on many infrastructure projects like these might be different if the full environmental costs were estimated and properly weighed in the decision-making process.

Significant expansion of deforested areas in Amazonia can be expected to decrease rainfall in the region, particularly in the dry season (Lean et al. 1996). Approximately 50% of the rainfall in Amazonia are derived from water recycled through the forest (Salati et al. 1979). This, when added to the effects of fire initiation foci and increased flammability through logging, can increase impact of an El Niño event of any given intensity.

Fragmentation

Fragmentation of forest into a landscape of small patches can lead to reduction of biodiversity through a wide variety of mechanisms (Laurance & Bierregaard 1997). The formation of edges causes a series of changes in microclimate that leads to death of large trees that provide most of the structure of forested ecosystems. A second set of effects is related to the size of the patches, which determines the viability of remaining isolated populations of different taxa, including ecological relationships among them such as dependence for food, pollination and seed dispersal. A third set of effects involves the distance between patches of forest, their relative sizes, and the ability of different kinds of organisms to cross the barriers separating the patches.

Depletion of fauna

Depletion of fauna occurs where large human populations live in proximity to the forest, depleting game species through hunting in accessible areas. These depletions provoke a series of impacts on the other taxa in the forest; impacts of faunal depletion have been particularly well-documented in tropical forests in Mexico (Dirzo & Miranda 1990).

Invasion by exotic species

Invasion by non-native species can have dramatic effects on forest ecosystems. Traditionally, the high diversity of tropical forests has been thought to protect these forests from the explosions of populations of invading species that characterize many simpler ecosystems. Despite their diversity, invasion by exotic species poses a significant threat in Amazonian ecosystems (Magnussen et al. 1998).

Climate change

Climate change is expected to have a wide variety of effects on Amazonian forests (Fearnside 1995). These include the effects of temperature increase from global warming, carbon dioxide concentration increase, rainfall changes both from global warming and from reduced evapotranspiration, extra-regional transport of smoke and dust, and increased cloudiness in some parts of the region. Extreme events are more important than changes in the means of variables such as precipitation and temperature. Climate change is likely to have its greatest impacts on Amazonian forests through its interactions with natural climate variability (such as that of El Niño), logging, fragmentation, and fires.

VALUE OF BIODIVERSITY

The purpose of valuation

Formidable obstacles, both in quantification and comprehension, have so far prevented valuation of tropical forests. The monetary value of forest products is the easiest form of value to calculate, but even this is not so easy to compute as it might seem. Problems include the discounting of future costs and benefits, and definition of who pays the costs and who receives the benefits that are included in the accounting. Assessments of other types of value have the additional difficulties that many potential uses of forest products are currently unknown and that money is inappropriate as an index for some of the most important types of value (Funtowicz & Ravetz 1994).

The purpose for which valuation is carried out is the prime determinant of what factors are appropriate to include. Only rarely would the objective of a valuation be to sell a natural ecosystem to a prospective buyer. One purpose of value calculations is for governments to use this information in making decisions about whether to take the sometimes painful decisions needed to stop destructive activities, such as deforestation in Brazilian Amazonia. Valuations for this purpose should be especially wide in scope because of government's responsibility as the representative of the interests of all Brazilians, including disadvantaged groups and minorities, and also future generations.

Another use of valuation is for setting the values of fines and of civil damage claims against those who destroy natural ecosystems. Although criminal and civil judgments are legally quite distinct, they share a common purpose, namely to deter would-be destroyers of ecosystems.

Financial value of utilitarian benefits

Factors Determining Financial Value

Financial value of the benefits of biodiversity is determined by various factors. Of great importance is the discount rate that is applied to translate future costs and benefits into current terms. Corporations and individual investors normally base financial decisions on discount rates determined by the rate that money can be earned in alternative investments in the economy. These rates are typically in the range of 10% yr^{-1} in real terms (i.e., after correction for inflation). Because discounting at these rates results in almost no weight being given to impacts or opportunities beyond about 30 years in the future, they are inappropriate for use by governments or others making decisions on the welfare of society at large, including the welfare of future generations. High discount rates also have perverse effects on sustainable use of natural populations, such as Amazonian rainforest trees (Fearnside 1989a). Differences in discount rates can have drastic effects on conclusions; choosing a discount rate of 3% versus a rate of 6% represents a difference of factor of 20 by the year 2100.

Irreplaceability is a characteristic of biodiversity that is not included in calculations that assume complete substitutability. Economic calculations generally assume that all things are interchangeable through the medium of money; unfortunately, this underlying assumption does not apply to many aspects of biodiversity. Biodiversity is not substitutable or interchangeable;

once a species or an ecosystem becomes extinct, there is no going back (Pires & Prance 1977). The permanence of extinction is a feature that provides an argument against discounting biodiversity benefits in the way that bankers in financial calculations routinely discount monetary values. This is different from arguments related to climate change, where carbon emissions can be reversed through uptake.

One aspect of the loss of biodiversity is the possibility of a Type II error, that is, of substantial negative consequences from assuming that there is no effect when, in fact, an effect exists. Most of science is devoted to eliminating Type I error, the chance of erroneously concluding that an effect exists when it does not, but the policy implications of Type II error may be greater in the case of biodiversity. The consequences would be permanent if effective action to reduce deforestation and assure continued presence of large areas of forest were not taken due to lack of knowledge of where thresholds lie for the areal extent of forest needed to maintain biodiversity.

By losing biodiversity one is losing ecosystem processes, not just the individual pieces of the ecosystem. An analogy has often been drawn between the loss of species and removing the rivets from an airplane's wing (e.g. Ehrlich & Ehrlich 1981). One may remove many rivets and see that the airplane continues to fly, but one day removing one more rivet can cause the wing to fall off and the airplane to crash to the ground. The danger of a crash is greatest if a species that is removed happens to represent a ‘keystone’ rivet. The need to conserve biodiversity is also indicated by the principle that one should ‘save all the pieces’ when tinkering with any unknown piece of machinery, as when dismantling a clock. If one throws pieces away, as one does by causing species extinctions, reassembling the machine becomes impossible.

When society makes decisions about the safety of different courses of action, it must be averse to catastrophic risk. For example, if one is considering the safety of nuclear power plants, one must have great certainty that the system and its components will not fail. The same applies to many of the changes being brought about by continuation of tropical deforestation (Fearnside 1997c). Each tree that falls increases the risk of dramatic changes, such as fire burning widespread areas of the remaining forest. The principal means of insurance against these risks is maintenance of large areas of unperturbed standing forest.

Presently-marketed forest products

The income that forests can produce continuously is one class of value that must be counted in any valuation scheme; it should not be confused with the monetary value of a one-time sale of everything contained in an ecosystem. The ability of standing forest to yield commercially-valuable products on a sustainable basis is, by itself, sufficient to make forest maintenance attractive when compared with the low-productivity cattle pasture that replaces forest in the Brazilian Amazon (Hecht 1992a). However, the high discount rates usually applied in assessing development projects in Amazonia can sharply reduce the appeal of any form of sustainable use (Fearnside 1989a). The obvious fact that vast tracts of forest are sacrificed for cattle pasture does not invalidate this, since the decision to convert forest to pasture is explained by the different factors involved. Thus, ranchers and speculators are unable and unwilling to make use of standing forest; felling it is, indeed, the most effective way for these recently-arrived groups to rid the area of the extractivists and indigenous peoples who previously inhabited it. Land speculation and government financial incentives add to the profitability of felling for pasture, even in the face of negligible production of beef (Fearnside 1980b, 1987; Browder 1988; Hecht et al. 1988; Hecht 1992b, 1993). Faminow (1998) has presented a contrary view; for a rebuttal, see Fearnside (1999a).

Wood extracted on long rotation cycles of sustainable forest management could produce significant income, although virtually no logging now practised in Amazonia is effected with sustainability in mind (Rankin 1985; Fearnside 1989a). Sustainable systems invariably require that the harvest be removed slowly. Leaving valuable timber in the forest would increase future returns dramatically, independent of its desirability from the standpoint of sustainability. Because the more commercially-valuable forests of Southeast Asia will soon cease to supply world timber markets, the price of timber relative to other commodities can be expected to rise significantly. Even if the forest were viewed as a resource to be mined unsustainably, Brazil would still be wise to leave the trees in the forest for a few more decades before cutting and selling them. A tree would be equivalent, for example, to a corporate bond with a 20- or 30-year maturity.

Biodiversity maintenance provides direct local benefits, such as providing non-timber forest products (Peters et al. 1989; Vásquez & Gentry 1989; Fearnside 1989b; Whitehead & Godoy 1991; Hecht 1992a; Richards 1993; Grimes et al. 1994; Pimentel et al. 1997a). Local benefits also accrue from the stock of genetic material of plants and animals needed to give a degree of adaptability to forest management and to agricultural systems that sacrifice biodiversity in nearby unprotected areas (Oldfield 1981; Myers 1989, 1992).

Presently-unexploited forest products

Many marketable products are either not exploited commercially or sold only in insignificant quantities. Native fruits represent one such class of products, with the exception of a few forest areas near urban markets. Peters et al. (1989) calculated that forest products (mainly fruits) from one hectare of forest near Iquitos, Peru, had a net present value of US$ 6,820, discounted to present value at 5% yr^{-1}. This cannot be generalized to Amazonia at large because the study site was only 30 km away from Amazonia's second-largest market for perishable local fruits.

Pharmaceutical compounds represent a value that is both virtually untouched and unrewarded by the present economic system (Mendelsohn & Balick 1995). Many drugs are first identified from natural compounds, and only later synthesized in laboratories where regularity of supply and uniformity of quality can be more easily guaranteed; there is also an issue of monopoly. While drug companies recognize the forest's usefulness (US House of Representatives 1983), they devote little attention to screening forest compounds because of the long lag time for gaining approval of new drugs (Farnsworth 1988). Indigenous people, who have accumulated knowledge of medicinal properties over centuries of use, can greatly facilitate identification of promising drug species (Balick & Mendelsohn 1992; Plotkin 1993). However, recent advances in automated screening technology have increased the ability of companies to simply analyse everything without guidance from traditional knowledge. Combinatorial techniques of generating vast numbers of compounds for testing have also weakened the demand for natural prototypes (e.g. Alper 1994).

An estimate of the opportunity cost of medicinal uses for rainforests in Mexico was US$ 6.4 ha^{-1} yr^{-1}, with a range from US$ 1 to US$ 90 (Adger et al. 1995). Substantial potential exists for identifying chemical models for pharmacological products based on Amazonian plants (Kaplan & Gottlieb 1990; Elisabetsky & Shanley 1994; Cordell 1995). However, both the role of biodiversity and that of traditional knowledge in making use of it are sometimes more limited than have sometimes been portrayed (cf. controversies regarding the rosy periwinkle of Madagascar: Djerassi 1992).

It is also important to realize that the financial value of pharmaceutical products, although clearly important, is not sufficiently large to support conservation on the scale sometimes envisioned (van Kooten 1998). In 1999, the barriers that the regulatory system in the USA pose to commercial development of pharmaceutical compounds derived from tropical biodiversity were made clear by the withdrawal of Shaman Pharmaceuticals from this field. This company had sent collecting teams to over 30 tropical countries, following a strict protocol to ensure that the value of any drugs discovered would benefit the local communities that had provided knowledge and material. Provir, an anti-diarrheal drug designed for AIDS patients, was nearing the end of its regulatory review when the US Food and Drug Administration ruled that an additional round of clinical trials would be required. The cost of the additional trials was beyond the company's resources, leading to the abandonment of all pharmaceutical work, including work on over 30 compounds considered promising for treatment of type II diabetes, which is untreatable with insulin). Shaman continues work on less-regulated products such as cosmetics and dietary supplements. Other companies, such as Merck, continue work on screening compounds from tropical biodiversity, but without any use of traditional knowledge from local peoples (Anonymous 1999b).

In addition, the amounts of money that can be obtained from pharmaceutical products are not likely to be very large, contrary to the expectations of some. An indication of this is the well-known Merck contract with Costa Rica's Institute of Biodiversity (Inbio), which provided an initial payment of US$ 1 million in cash plus US$ 135,000 worth of technology for refining samples, followed by an undisclosed percentage of future royalties believed to be in the range of 1-3% (Crook & Clapp 1998, p. 137). The arrangement is in exchange for samples collected in the country's natural ecosystems by a network of parataxonomists (Roberts 1992). This flow of samples is sufficient to satisfy Merck's capacity to invest in searching for new compounds from tropical forests. It is important to realize that Brazil is in competition with Costa Rica and the rest of the tropics, and that the limiting factors to gaining income from biodiversity in this way are laboratories and taxonomists, not forests and tribal peoples.

Genetic materials represent another class of value that has little present use and even less financial return for local inhabitants. Several present plantation species, such as cacao and rubber, are native to Amazonia. Resistant varieties will become indispensable when pathogens such as Microcyclis ulei in rubber and Crinipellis palmivora in cacao eventually reach the plantation areas to which these species have been taken (Oldfield 1981). Southeast Asia in the case of rubber, and West Africa in the case of cacao, are presently protected by their distance from the crops' land of origin (Ampuero 1977; Holliday 1977). Many other forest species could undoubtedly be useful to humans.

Monetary value of environmental benefits

Amazonian forest has many ecological functions that are rarely taken into account in assessing monetary value. These functions include protecting the soil from erosion, thereby protecting navigable rivers and hydroelectric reservoirs from siltation. Measurements of erosion in 10 m X 10 m plots in Ouro Preto do Oeste (Rondônia), Manaus (Amazonas), Apiaú (Roraima) and Altamira (Pará) indicated much more erosion under pasture than under forest at all locations (see Fearnside 1989c). Measurements of erosion using arrays of stakes indicate approximately 1 cm yr^{-1} of soil loss under annual crops such as rice and maize, in contrast to minimal erosion under forest (Fearnside 1980c).

The role of the forest in maintaining the regional water cycle is of major economic importance. The potential magnitude of change that could be caused by large-scale replacement of Amazonian forest with cattle pasture is indicated by the soil erosion plots mentioned above, where runoff is approximately ten times greater under pasture (Fearnside 1989c). Water that runs off on the surface rather than sinking into the soil cannot be sucked up by the roots of trees for return to the atmosphere via evapotranspiration. An appreciable amount of the rain in Brazil's principal agricultural areas in the central-southern part of the country derives from the Amazon forest (Salati & Vose 1984). Increased runoff expected after removing the forest would greatly alter aquatic habitats in the region, and would interfere with human use of the várzea (white-water floodplain), which is the region's richest agricultural resource (see Fearnside 1985a). Maintenance of a substantial but poorly-quantified fraction of the forest is necessary to prevent a changed rainfall regime from degrading and eventually destroying the remainder of the forest. No simple cutoff point exists beyond which further clearing would lead to climatic catastrophe (Fearnside 1997c); rather, every tree that falls increases the probability that the addition to the already very large natural variation in rainfall will provoke an unprecedented drought, setting in motion a destructive positive feedback process (Fearnside 1985b).

Avoidance of global warming is the benefit of forest maintenance for which the world is most willing to pay at present (Pearce 1998). Each hectare of forest clearing in 1990 released 194 t of carbon (C) as a net committed emission, in other words, after deducting the future regrowth of secondary forest and other components of the replacement landscape (Fearnside

1999b, updated from Fearnside 1997d). Considering this release per hectare and a market price for avoided emissions that can be credited against the allowable amounts of emissions under the Kyoto Protocol of US$ 5-35 t^{-1} C (see Fearnside 1999c), the value of avoiding deforestation corresponds to US$ 970-6790 ha^{-1} of forest, with a midpoint of US$ 3880 ha^{-1}.

The contrast of these values with current returns from cutting the forest is clear. The price of forested land in Brazilian Amazonia averaged approximately US$ 150 ha^{-1} over the 1997-1998 period. Much land is available for as little as US$ 30 ha^{-1} (Santiago 1999). Although purchasing land is not proposed, the price of land is important as an indicator of what it can produce under the use options currently open to buyers, that is of selling the timber and converting the land to cattle pasture. The land price represents the net present value (NPV) of the income stream from deforestation, considering the discount rate employed by investors in their financial decisions. The value of the carbon benefits from keeping the forest are 5-45 times higher than the value of deforestation, while the value of the 1998 deforestation was US$ 1.6-11.4 billion (see Fearnside 1999c).

Non-monetary values

Non-monetary values of rainforest are key components of human decisions about the future of these ecosystems. While Amazonian forests are worth a very large amount of money, it is not for this reason that people are so concerned about their fate. After all, oil fields and coal deposits are also worth tremendous amounts of money but do not inspire concern beyond that derived from their direct use value and the impacts of pollution and climate change provoked by burning them.

Tropical rainforests teem with life. They are also home to unique and highly-threatened human cultures (Ricardo 1996). Many people around the world hold the view that indigenous peoples and biodiversity are not things that modern society should simply extinguish if doing so should prove profitable (Wilson 1984; Kellert 1993). This viewpoint is not based on financial calculations.

Non-economic and non-utilitarian arguments for maintaining natural habitats have been presented by many authors (Budowski 1976; Ehrenfeld 1976; Jacobs 1980; Janzen 1986; Poore 1976; Wilson 1992). E.O. Wilson summarized this well by his description of destruction of biodiversity as ‘the folly that our descendants are least likely to forgive us’.

While a growing number of people recognize the non-monetary value of biodiversity, a very substantial number of people do not. It is pure fantasy to think that one could walk up to someone who is holding a chainsaw and, with an arm around the person's shoulder, make a convincing argument that biodiversity is more important than cutting down the tree. It is important to understand that, in order to reduce biodiversity loss, those who believe biodiversity to be important need not convince those who do not hold this belief to change their position. It is sufficient for such non-believers to understand that a substantial number of people in the world do believe that maintaining biodiversity is important, and that this translates into a potential monetary flow to achieve that end. (e.g. Ruitenbeek 1992; Pearce 1998). This 'willingness to pay' can influence events, independently of the opinions of potential deforesters on the importance of biodiversity. Types of value of Amazonian forests are summarized in Table 1.

[Table 1 here]

Willingness to pay and willingness to accept

Many benefits of biodiversity are global rather than local (Swanson 1997, pp. 76-78). The stock of useful chemical compounds, and of genetic materials for other than local use, represents an investment in protecting future generations in distant places from the consequences of lacking that material when it is needed one day. This value is different from the commercial value of products that may be marketed in the future, which would represent a lost local opportunity should biodiversity be destroyed. A medicinal use, such as a cure for a disease, is worth more to humanity than the money that can be earned from selling the drug.

The value of biodiversity is poorly quantified, and severe methodological constraints limit our ability to assign meaningful monetary values to it (Norton 1988; Stirling 1993). While one knows that its monetary value is very high (Pearce & Moran 1994; Costanza et al. 1997; Meijerink 1995; Pimentel et al. 1997b), the willingness of the world at large to pay is the factor limiting how much of this value can be translated into a monetary flow. That willingness to pay has, in general, been increasing, and it may increase substantially in the future (Cartwright 1985).

Many of the values of ecosystems are not marketed in today's human economy, and therefore often receive little weight when business and political decisions are made (e.g. Foy 1990). Existence value is something that accrues mostly to populations who are either very close to the forest, such as indigenous peoples, or who are far removed from it, such as urban dwellers elsewhere. Non-market values include ethical, cultural and scientific values for the current human population, as well as utilitarian functions of the ecosystems, including as yet undiscovered functions, for future generations. Non-marketed values of environmental services are often assessed through contingent valuation techniques such as willingness to pay (WTP) for keeping the service and willingness to accept (WTA) its loss (Johansson 1990; Carson 1998). It should be emphasized that the monetary values generated by such techniques are not real values, and that the people most interested in maintaining natural ecosystems often cannot afford to pay anything (e.g. Attfield 1998). Nevertheless, WTP does provide an indication of the scale of monetary flows that might one day be tapped to prevent loss of natural ecosystems and support their inhabitants on a sustainable basis. In the case of Amazonian forest, the value of these environmental services greatly exceeds the income that can lastingly be earned by deforesting the land (Fearnside 1997e).

Penalties for destruction of biodiversity

In order to deter violators, the penalties must be neither too high nor too low. If they are set too high, for example, assessing forest at US$ 1 billion per hectare, then fines or damages will simply never be collected. If penalties are too low, as is much more often the case, then violators will simply pay the fines and continue to destroy forest. This has happened on numerous occasions.

Intimately associated with the value of penalties is the perceived likelihood that they will be collected. The usual practice in decision-making is the Bayesian calculation of expected monetary value (EMV) (e.g. Raiffa 1968), which is the sum of the products of all possible monetary outcomes, multiplied by their respective probabilities of occurrence. If the probability of being forced to pay the penalties is near zero, then the value of the fine and/or damages would have to approach infinity in order to make compliance with the law financially rational. This possibility is evident in the case of efforts of the Brazilian Institute for the Environment and Renewable Natural Resources (IBAMA) to apply fines to those burning Amazonian forest without permits. Each year since 1989, IBAMA has issued fines with nominal values totaling the equivalent of many millions of dollars, yet only a small fraction (averaging 6% in 1997) have been collected. The Environmental Crimes Law (Decree Law No. 9605 of 12 February 1998) is expected to increase the percentage of fines collected, although recent setbacks have delayed implementation of most provisions of the law (Gonçalves 1998, ISA 1999).

In addition to the probability that violators will eventually have to pay at all, the time likely to elapse between violation and payment is also important. Delays diminish the deterrent effect, independent of the correction for inflation. Money in hand can be invested while a judicial process drags on; the delay is similar to having an interest-free bank loan. The discount rates used by the infractors themselves will be the key factor in determining the little weight given to a fine or damage claim to be paid years in the future. It is therefore essential that the judicial system be both strengthened and streamlined; setting high values for penalties is, in itself, not sufficient to deter environmental destruction.

The present combination of low values for penalties, low probability of collection and long delays, make environmental legislation ineffective in altering the behaviour of those who destroy natural ecosystems. Those contemplating destroying an ecosystem can be expected to compare the immediate financial gains that they would forego by complying with environmental legislation to the discounted value of the fines and damage claims that would arise from violations, multiplied by their respective probabilities of being collected in practice.

It is worth noting that the threat of jail sentences has little effect in spite of ‘ecological crime’ having been created in Brazil’s 1988 constitution as an offense punishable by imprisonment. Because many of the larger aggressions against the environment are committed by corporations or by wealthy individuals, the applicability of imprisonment is limited, because Brazilian law makes it almost impossible to imprison anyone with a university education and ‘good antecedents’. The Environmental Crimes Law softens the penalties that would be applied to corporate executives, obliging them to work on community projects, thereby increasing the probability that the penalties will actually be imposed.

In order to resist the efforts of violators and their attorneys to contest fines and claims that courts award, values must be well-grounded in scientific terms. It is important to bear in mind, however, that the true value of the natural ecosystems is almost invariably much greater than figures reflected in judicial decisions. How high the value chosen is depends on how far down the list of losses and impacts one goes, that is, how willing one is to include types of impacts for which quantification is less certain. Values should be chosen with deterrence in mind, meaning that they should be set as high as possible, without becoming counterproductive by inhibiting application in practice.

Setting values with deterrence in mind means, to a certain extent, that the procedure is not scientific. In science, one must approach a problem without a priori biases, and accept whatever conclusion the experimental or other results indicate. In the case of establishing a value for forest, however, one already knows the conclusion beforehand, namely that cutting the forest is undesirable and should be deterred. If financial calculations indicate otherwise, then the conclusion is not that the forest should be sacrificed, but rather that the financial formula is wrong and should be modified. This is the same situation that applies to calculations by investors that, under currently used procedures, often lead to financially 'rational' decisions to destroy potentially-renewable natural resources like forests (Fearnside 1989a). It is a situation similar to that of an Agatha Christie detective novel; using a brilliant but judicially-unacceptable line of reasoning, the detective discovers who has committed a crime, after which the task of amassing evidence for the trial is turned over to the plodding and unimaginative police. In this case, valuation is needed for both phases: the true value (including many poorly-known and nonmonetary factors) is needed to decide that forest destruction must be deterred, and one must also have valuations for building a strong legal case with monetary penalties that cannot be challenged. Successively more types of value can be expected to be incorporated into the latter category as further progress is made in quantifying values currently considered too vague for use in legal proceedings. Progress has been made in developing methods for establishing such values (e.g. Gregory 1986; Pearce 1999).

One criterion often applied is replacement value. In the case of natural ecosystems, however, it is frequently assumed from the outset that replacement is impossible, meaning that other criteria are used that give costs within an acceptable range. Replacement with the same forest is, of course, impossible. Nevertheless, an approximation of the original natural ecosystem can be had if one is willing to pay for it. The quality of the replacement can vary tremendously, and becomes much more expensive as more of the original functions of the natural ecosystems are included. Although examples of restored lowland humid rainforest are nonexistent, an idea can be had from the project led by Janzen to restore a portion of the tropical dry forest in Costa Rica's 50,000-ha Guanacaste National Park (Janzen 1988). This project has a budget of over US$ 50 million, and depends on something that cannot be bought, namely the dedication of biologists of the calibre of Janzen to do the research necessary to decide how funds can best be applied to restore the myriad ecological interactions of the forest.

Restoration ecology is in its infancy for Amazonian forest. One start, on a limited scale, is the programme of Mineração Rio do Norte to re-establish forest on areas it has mined for bauxite at Porto Trombetas, Pará, where the cost of establishing a secondary forest on the inhospitable areas left after bauxite mining is US$ 6000-7000 ha^{-1} (João Ferraz, personal communication 1999). This expense can be easily paid by the mining company because of the high value of the ore (US$ 27 t^{-1}, or about US$ 5 million ha^{-1}), but this is a situation that does not apply to most activities destroying Amazonian forest. In addition to financial cost, it is important to remember that this project also depends heavily on having someone with a tremendous amount of experience with Amazonian forestry, namely in this case Henry Oliver Knowles personally devoting himself to the task. Such individuals are not so readily available for hire as are chainsaw operators or bulldozer drivers, and this severely limits the practicality of widespread restoration of Amazonian forests.

The type of restoration that far outweighs all others in Brazilian Amazonia is the recuperation of forest vegetation on degraded pastures. This has not been done, although much information on successional processes has been collected with this goal in mind (e.g. Nepstad et al. 1991; Uhl et al. 1991; Gascon & Moutinho 1998).

One of the factors that must be taken into account in establishing the cost of restoration is the time allowed for replacement ecosystems to become established. If infinite time is available, then restoration can be had for free by simply abandoning sites and waiting for natural seed dispersal and successional processes to take place. When more speed is required, restoration becomes increasingly more expensive. It is therefore essential to establish the discount rate applied in calculating the value of time spent in effecting the restoration.

CONSERVATION OF BIODIVERSITY

Types of protected areas

Protected areas in Brazil are created under different legal provisions, and have varying degrees of restriction on the use of the areas involved. National parks, biological reserves and ecological stations are all administered by IBAMA. In the case of ecological stations, a small fraction of the station may be cleared for experiments. State governments have created reserves, such as the Mamirauá 'Sustainable Development' Reserve in Amazonas, where the local population remains in the area and is given responsibility for managing it in accord with a zoning plan that includes a portion to be left untouched. IBAMA creates 'areas of environmental protection' (APAs), where populations, including towns, are included; development restrictions apply within the areas, but the result is less than what would be expected, for example, of a conservation unit as defined by the World Conservation Union (IUCN). IBAMA also authorizes extractive reserves, where traditional residents extracting rubber, Brazilnuts and other non-timber forest products (NTFPs) are given use rights to the land in exchange for their commitment to protect the forest (see Allegretti 1990, 1994). Extractive reserves created by IBAMA are not to be confused with 'agro-extractive settlement projects' (PAEs) created by the National Institute for Colonization and Agrarian Reform (INCRA), or with plans of the Amazonas state government for 'ecological' settlements where extractivism is supposed to be practised by large numbers of people who lack an extractivist background. Other kinds of units include research reserves administered by institutions such as the National Institute for Research in the Amazon (INPA) and the Emilio Goeldi Museum (MPEG), and private reserves (Áreas de Relevante Interesse Ecológico).

Indigenous reserves, which are administered by the National Indian Foundation (FUNAI), cover by far the greatest extent of forest. However, these cannot be considered protected, as the tribes may, in the future, decide to adopt land uses that destroy forest (Fearnside & Ferraz 1995). Nevertheless, up to the present, indigenous peoples have had the best record of maintaining forest intact, and in many parts of the region the only forest still standing is that in indigenous reserves. Negotiations are needed with the indigenous peoples involved to ensure that they maintain their forests intact, and that they receive benefits from the environmental services these forests provide (Fearnside 1997e; Fearnside & Ferraz 1995).

Status of protected areas

The areal extent of protected areas is still small relative to the extent of Amazonian forests. Brazil's 1998 endorsement of the World Wide Fund for Nature (WWF) 'Forests for Life' programme goal of protecting 10% of its forests has been set back by attempts by IBAMA to claim national forests, which are set aside for timber exploitation rather than for environmental protection, as part of the country's achievement in this area.

The types of ecosystems protected in the current system of conservation units are extremely uneven. Of 111 'vegetation zones' in Brazil's Legal Amazon region as defined by IBAMA vegetation types (Brazil, IBGE & IBDF 1988) occurring within each state, only 37 (33%) had any portion included in a conservation unit by 1990 (Fearnside & Ferraz 1995).

The level of disturbance varies greatly among different conservation units. Illegal logging is frequent; in some cases invasion by ranchers and farmers also occurs (Cotton & Romine 1999). Brazil has a poor record of extinguishing parks or building roads through them when development proponents find these attractive (Fearnside & Ferreira 1985). Eighty-five percent of Brazilian conservation units are not effectively implemented (Ferreira et al. 1999).

'Paper parks', or conservation units that are decreed and drawn on a map but have little or no on-the-ground implementation are common in Brazilian Amazonia. As unfortunate as this situation is, a rationale does exist for rapid creation of paper parks because opportunities for park creation are likely to decrease substantially in the future as Amazonian forest land becomes more expensive and as areas without significant human occupation diminish. Since the park decree rate has been much higher than the park loss rate, the result has been a net gain for parks by investing in a strategy of maximizing new park creation, as opposed to one of strengthening defense of existing parks. Eventually a transition will have to occur, with greater attention being given to consolidating existing parks.

People in reserves

Brazil has been a centre of debate on the question of whether conservation units should be designed and managed to include people living in the reserves (e.g. Kramer et al. 1997). A 'fortress approach', whereby uninhabited reserves are guarded against encroachment by a hostile population in the surrounding area, is believed to be unworkable as a means of protecting biodiversity, in addition to causing injustices for many of the human populations involved. Brazil has a proposed National System of Conservation Units (SNUC) that would require arrangements to allow continued presence of residents in a variety of types of conservation units.

The balance of advantages and disadvantages of having humans living in reserves varies with each situation. Impact of human presence can be substantial through hunting and agricultural clearing (e.g. Redford & Stearman 1993; Peres 1996, nd).

A key question is whether logging is to be allowed in extractive reserves. The fall in rubber prices since the first extractive reserve was created in 1988 has left rubber tappers in economic straits. Rubber tappers are divided as to whether timber extraction should be allowed. Arguments against opening these reserves to timber extraction are that logging is fundamentally different from non-timber forest product (NTFP) extraction; while rubber has been tapped for over a century without significantly damaging the forest, the virtually-universal experience with logging has been the opposite, namely that it is neither sustainable nor environmentally benign. Although proposals to open these reserves are invariably presented as 'experiments' with sustainable management, in fact they are not experiments in the social context of extractive reserves. In an experiment, the result can go either way; the system under test may work or it may not. If it proves unsustainable, the assumption is that the experiment would be stopped and the system not implemented. However, extractive reserves are inhabited by human beings rather than laboratory mice, and terminating an unsuccessful 'experiment' is not automatic. Selling wood in addition to NTFPs inevitably produces more cash income than NTFPs alone. Once people become accustomed to receiving a higher income than they do from NTFPs alone, they will not want to go back to their former level of subsistence should researchers tell them that their timber management is not sustainable.

An additional problem is that the first cycle or two of any timber management system virtually always yields more than subsequent cycles because the large trees that have been growing for centuries are there for the taking, whereas eventually the management system will only be able to harvest what has grown while the managers wait. The transition to the sustainable equilibrium harvest level therefore implies a reduction of offtake, and consequently of income, which people may not be willing to accept. If they persist in higher levels of offtake, the system will be unsustainable. In this context it is important to emphasize the distinction between extractive reserves created by IBAMA and agro-extractive settlements created by

INCRA. The former are created with the express purpose of protecting the environment, while the latter are to absorb migrant population and produce commodities (Brazil, INCRA 1996). Timber management underway in the Porto Dias extractive settlement area in Acre may produce information on the viability of community management of timber, but it should not be viewed as a precedent for opening extractive reserves to logging.

Buffer zones

Buffer zones (áreas de entorno) are a key part of the design of any conservation unit (Sayer 1991). The human population living around the reserve must have a sustainable livelihood if it is not to invade the reserve. To this end, a number of efforts have been made to aid surrounding populations in developing means of support that do not involve invading the reserve. Wells and Brandon (1993) reviewed 23 such projects designed to promote community involvement around the world, and found discouraging results. A number of lessons can be learned concerning what types of projects are most likely to succeed (see Perrings et al. 1995, p. 892). It is important that local people get concrete benefits from biodiversity protection, or they will inevitably destroy these most valuable resources (Dove 1993). An important caveat is that the means they employ must be sustainable (Foy 1990). Population size must remain within the limits of carrying capacity as a prerequisite for sustainability (Fearnside 1997f).

Choice and design of reserves

The choice and design of reserves is invariably based on both biological and practical criteria. Biological considerations include the diversity of species present at a site, and the need for representation of the different natural ecosystem types within the protected area system. A series of maps and recommendations were compiled with this end by a major workshop held in Manaus (Rylands 1990). The extent of biological knowledge for different locations is an important factor that works in both directions in the criteria adopted by the workshop; well-studied sites are found to be highly diverse precisely because they have been well studied (Nelson et al. 1990), while poorly-studied sites receive additional weight because they are unknown.

Endemism and the presence of species with very restricted ranges provide a rationale for assigning priority to different locations. In the case of birds, these criteria make Andean sites and Atlantic forest more urgent priorities than forests in Brazilian Amazonia (Fjeldsa & Rahbek 1997). Minimum critical size, and the relative merits of ‘one large versus several small’ reserves, could also influence decisions, but matters of practicality are usually more telling in decisions on reserve creation (e.g. Foresta 1991). Other concerns include location to maximize vegetation-type representation (Fearnside & Ferraz 1995), and location for maximum defensibility of reserve boundaries (Peres & Terborgh 1995). Reserve creation is often highly opportunistic, and decisions must be taken quickly when opportunities arise. Cost has often been a dominant factor. It is much cheaper to create reserves in sparsely-populated areas of the state of Amazonas than it is near the deforestation frontier in, for example, Mato Grosso. Not only is the price of land much higher near the deforestation frontier, but the cost of defending the reserves is also very much higher. The result is that the state of Mato Grosso has very little protected area, despite high levels of endemism in the ecotones between the forest and cerrado [central Brazilian scrub savanna] domains.

ENVIRONMENTAL SERVICES AND FUTURE PROSPECTS

The capture of value from environmental services represents a long-term prospect for sustainable support of the human population in the Amazonian interior, but many obstacles remain to realizing this potential (Fearnside 1997e). Biodiversity maintenance is one of the many services provided by Amazonian forests. This service can by captured jointly with other services, such as climate regulation. Negotiations under the United Nations Framework Convention on Climate Change (UN-FCCC) are currently much more advanced than those under the biodiversity convention, raising the possibility that significant monetary flows could begin within a decade for global warming mitigation (Fearnside 1999d). If avoidance of deforestation is included in these measures, it will also bring benefits for biodiversity protection. Deforestation can be avoided through changing policies that affect such factors as land speculation, land-tenure arrangements, road construction and settlement policy. It can also be avoided through reserve establishment and protection. In the area of reserve protection, the key to protecting large areas lies in involvement of the region's indigenous peoples. These and other traditional residents of the region must have a role as full partners in efforts to sustain environmental services, including Amazonian biodiversity.

CONCLUSIONS

Biodiversity contributes to making Amazonian forests very valuable, leading to the conclusion that they must be protected. This protection must not await better data on valuation. Protecting Amazonian forests requires understanding of the process of deforestation, and changing policies such that actors are motivated to maintain forest rather than cut it down. No matter how severe, penalties are no substitute for removing the motives for deforestation. Many effective measures could be taken through government action. These include levying and collecting taxes that discourage land speculation, changing land-tenure establishment procedures so as not to reward deforestation, revoking remaining incentives, restricting road building and improvement, strengthening requirements for environmental impact statements (RIMAs) for proposed development projects, and creating employment alternatives.

Decisions regarding creation and management of various kinds of reserves are likely to be key factors in determining the long-term future of biodiversity in Brazilian Amazonia. Rapid creation of ‘paper parks’ has been an effective strategy to increase the area in conservation units while this is still financially and politically feasible, but emphasis must eventually shift to protection of already-established parks. Agreements with traditional populations inhabiting forest areas, especially indigenous peoples, offer the greatest potential for maintaining large areas of forest with their biodiversity relatively intact. Mechanisms are needed to insure that the value of biodiversity maintenance and other environmental services provided by these forests accrues to the traditional peoples who defend them.

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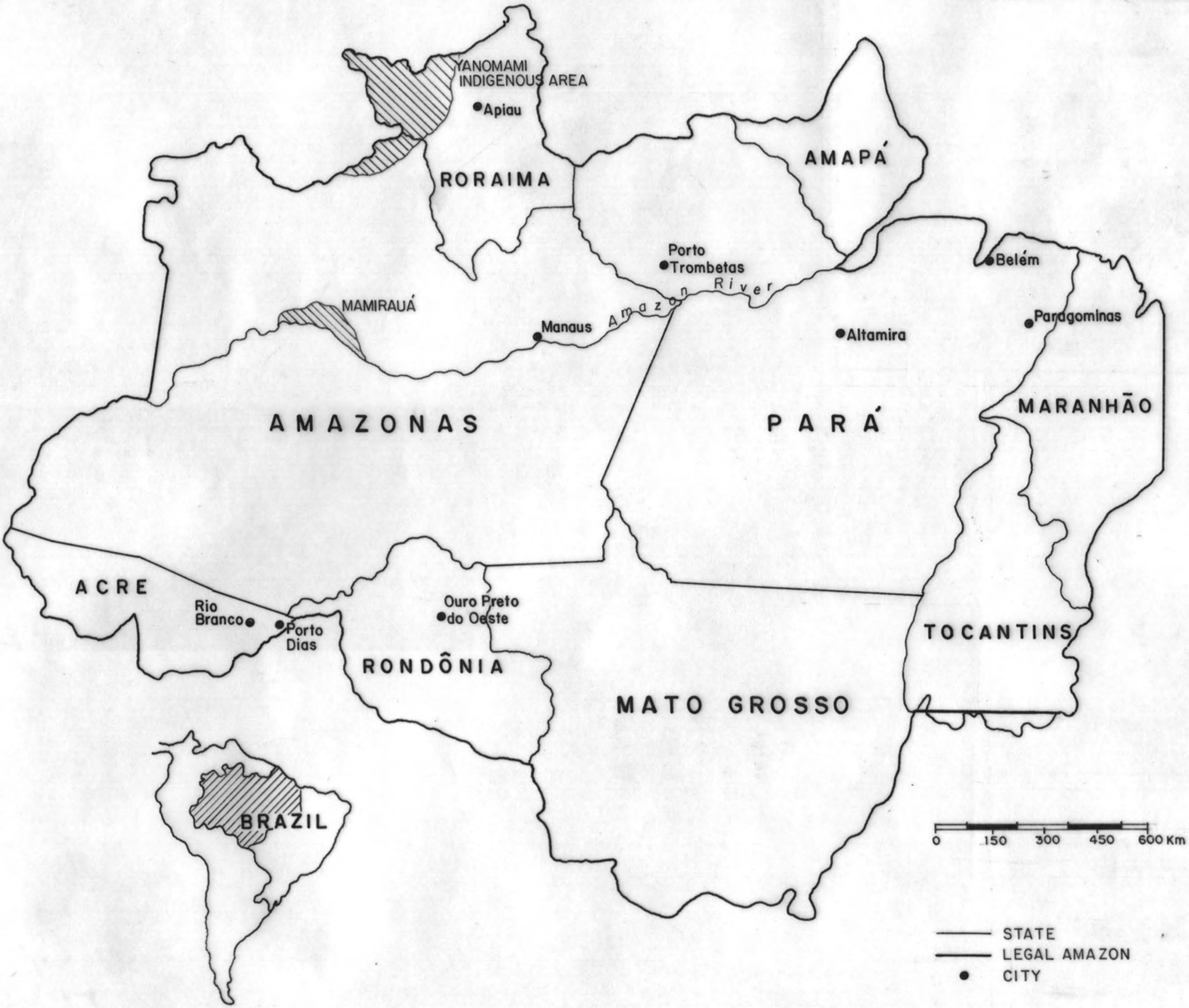
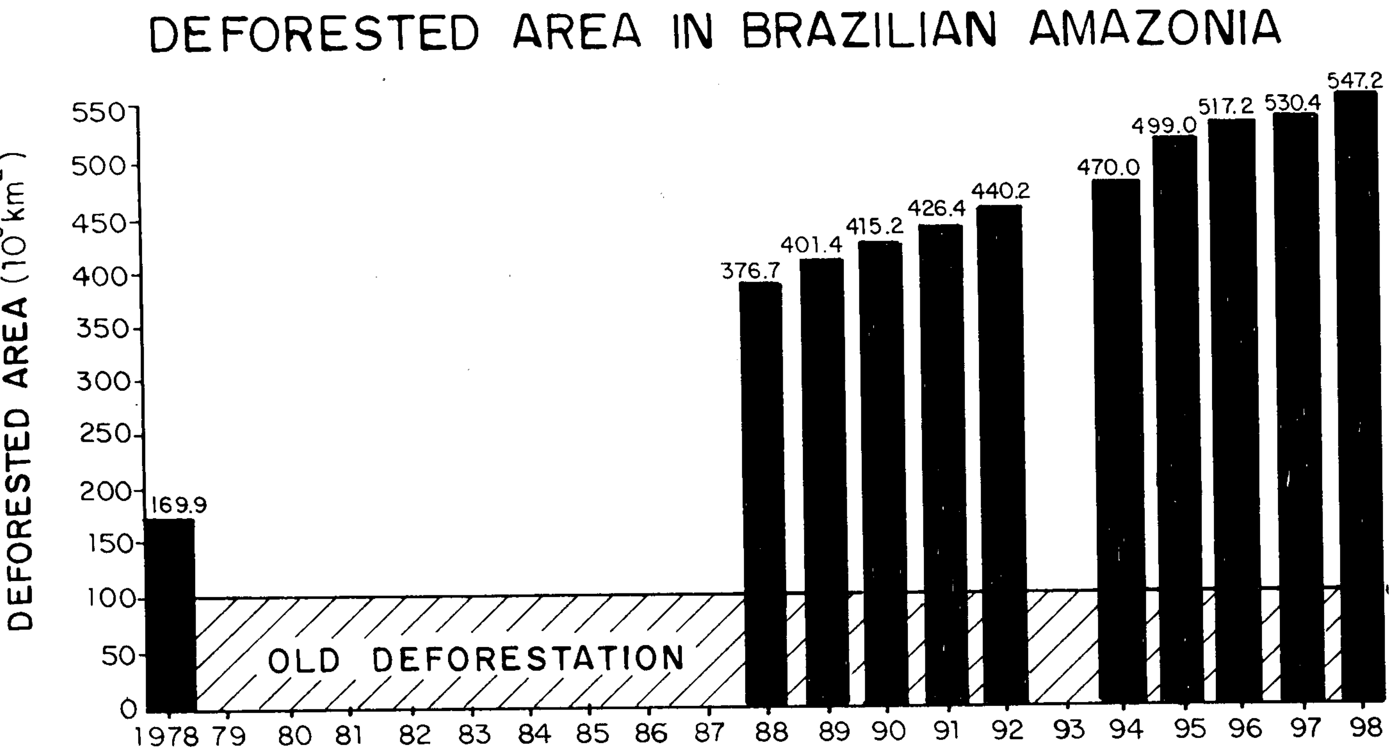
FIGURE LEGENDS

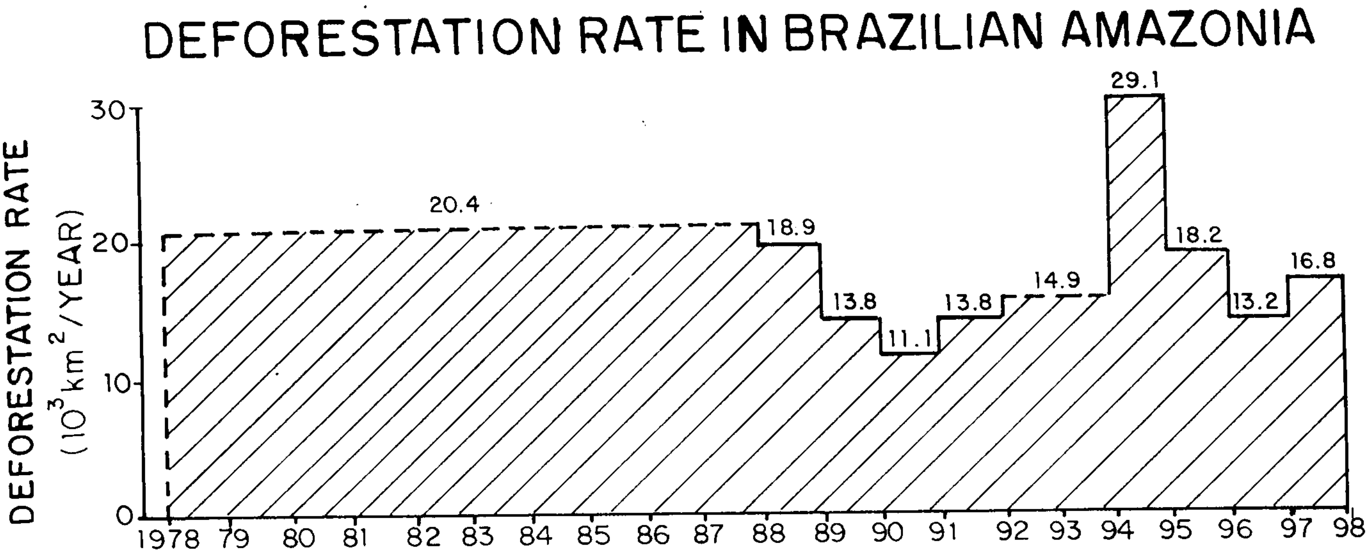
Fig. 1 – Brazil's Legal Amazon region with locations mentioned in the text.

Fig. 2 – Extent (A) and rate (B) of deforestation in the Brazilian Amazon (Fearnside 1997a; Brazil, INPE 1998, 1999).

Table 1: Types of Value of Amazonian Forests	
Type of value	Examples
Commercial value of presently-marketed commodities	Timber, beef
Commercial value of commodities not presently marketed	Pharmaceutical products
Environmental services	Carbon storage, water cycling
Existence values	Non-utilitarian value of biodiversity and cultural diversity


YANOMAMI INDIGENOUS AREA
Apiau
RORAIMA
AMAPÁ
Porto Trombetas
Belém
MAMIRAUÁ
Manaus
Amazon River
Altamira
Paragominas
AMAZONAS
PARÁ
MARANHÃO
ACRE
Rio Branco
Porto Dias
Ouro Preto do Oeste
RONDÔNIA
TOCANTINS
MATO GROSSO
BRAZIL
0
150
300
450
600 Km
STATE
LEGAL AMAZON
CITY


DEFORESTED AREA IN BRAZILIAN AMAZONIA
DEFORESTED AREA (10^3km^2)
169.9
376.7
401.4
415.2
426.4
440.2
470.0
499.0
517.2
530.4
547.2
OLD DEFORESTATION
1978 79 80 81 82 83 84 85 86 87 88 89 90 91 92 93 94 95 96 97 98


DEFORESTATION RATE IN BRAZILIAN AMAZONIA
DEFORESTATION RATE (10^3km^2/YEAR)
20.4
18.9
13.8
11.1
13.8
14.9
29.1
18.2
13.2
16.8
1978 79 80 81 82 83 84 85 86 87 88 89 90 91 92 93 94 95 96 97 98