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Is there too much or too little natural forest in the Atlantic Zone of Costa Rica?

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Abstract: Deforestation rates in developing countries are often regarded as excessive, despite the lack of a satisfactory economic benchmark to evaluate this claim. This paper provides such a benchmark for a particular region in Costa Rica. The monetary value of the various functions performed by tropical rainforests is estimated and used in a conventional optimal control model to compute the globally optimal natural forest stock in the Atlantic Zone of Costa Rica. The results indicate that the current forest stock is suboptimally large, suggesting that promoting further forest conversion can increase economic welfare. The current stock would be near optimal only when (*i*) the annual benefits of carbon fixation are extremely high and (*ii*) the government of Costa Rica would be fully compensated for this positive externality.

Résumé : Malgré l'absence de points de repères satisfaisants de nature économique pour tester cette affirmation, on considère souvent que le taux de déforestation dans les pays en voie de développement est excessif. Cet article fournit un tel point de repère pour une région particulière du Costa Rica. La valeur monétaire des différentes fonctions de la forêt ombrophile tropicale a été estimée. Ces valeurs sont utilisées dans un modèle conventionnel de contrôle optimal pour calculer globalement le stock optimal de forêts naturelles dans la zone atlantique du Costa Rica. Les résultats montrent que le stock actuel de forêts est sous-optimal et trop élevé, ce qui suggère qu'on pourrait améliorer le bien-être économique en encourageant davantage la conversion de la forêt. Le stock courant sera à peu près optimal lorsque (*i*) les bénéfices annuels dus à la fixation du carbone seront extrêmement élevés et (*ii*) que le gouvernement du Costa Rica sera pleinement compensé pour ces retombées bénéfiques.

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1. Introduction

Tropical deforestation is a prominent environmental issue that has attracted widespread attention in recent years. Deforestation is often attributed to market failure and (or) policy (government) failure. Market failure occurs when benefits from preservation of tropical forests, such as their contribution to global climate regulation, spill over to other countries that do not compensate the forest owner for these services. Policy failure in the context of tropical forest management occurs when taxes and subsidies deliberately or inadvertently distort the level playing field. The latter issue has been studied extensively in the 1980s and 1990s (e.g., Brown and Pearce 1994; Repetto and Gillis 1988). In many studies, market and government failure are brought forward to explain deforestation patterns that are socially excessive and wasteful.

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However, the existence of market- and policy failure leaves undisputed that (some) forest conversion may be economically rational and in the interest of society at large. Indeed, this would be the case when the net discounted benefits of alternative land-use options exceed those of sustainable forest management or strict preservation, when properly valued. While many authors are sceptical about how forests are being valued and argue that current deforestation is excessive (e.g., Repetto and Gillis 1988; Reed 1992; Pearce and Warford 1993; Barbier et al. 1994), in the absence of empirical studies that aim to incorporate the full value of standing forests and alternative land uses, this remains an issue open for dispute.

Given the amount of work done in the analysis of government (and market) failure and the progress made in these fields, surprisingly little is known about "socially optimal deforestation" or "socially optimal forest stocks." This basically implies that the benchmark against which deforestation should be measured is currently lacking.

This paper analyses whether the currently remaining natural forest stock in the Atlantic Zone (AZ) of Costa Rica is economically optimal or not. Economic optimality is determined from a "global" perspective, implying that domestic and transboundary benefits of forest conservation are considered. The AZ region, as defined in this study, is confined to the area north of the Talamancan mountain range, running north of the highway San José – Limon, and from the

Braulio Carillo National Park in the west to the Atlantic coast in the east. In total it comprises an area of about 545 000 ha. The natural vegetation in the study area is classified as humid tropical forest.[2] Currently, the forest stock in this zone amounts to approximately 150 000 ha (Stoorvogel and Eppink 1995), most of which is located in protected areas.

From an economic viewpoint, forest preservation is only one of many land-use options. The socially optimal land-use pattern is one that maximizes the present value of welfare for society at large. Hence, at the margin, the returns from sustainable forest management should be competitive with those of alternative land uses, which may include agriculture and mining. In this respect it should be recognised that the benefits of sustainable forestry include not only marketable products obtained from forest management (such as timber and nontimber forest products) but also various regulatory and habitat functions for which market data are typically not readily available.

In this paper we try to include estimates of the monetary value of all the major functions (marketable and nonmarketable) performed by natural forests. Even though Ehui and Hertel (1989) have broken new ground by computing the "optimum" forest stock for Côte d'Ivoire with an optimal control model, their analysis was based on financial returns of forestry and the competing land use (agriculture), rather than a measure of the economic welfare generated by forests as a standing stock. Since their model did not include positive externalities, such as preservation of genetic diversity and climatic benefits, the optimum steady-state forest stock is underestimated. The economically optimal forest stock in this paper is defined as the total size of forest cover that maximizes the present value of social benefits.[3]

An important contribution of the study is that we demonstrate the relevance of allowing for site-specific characteristics that enable a valuation of the agricultural frontier at the margin, as opposed to the approach of Ehui and Hertel (1989), which basically measures average productivity. Data with respect to agricultural productivity at the margin in the AZ of Costa Rica has been gathered by intensive fieldwork during the period 1987–1997 (e.g., Bouma et al. 1995). The analysis is deterministic, and this is certainly a simplification. For example, it is argued in economic literature that harvesting nontimber forest products takes place as an insurance for agricultural crop failure (e.g., Baland and Platteau 1996), suggesting that attitudes to risk will partly determine the optimal forest stock that society should retain. Similarly, it has been observed that some landowners in the region aim to diversify their production and, therefore, may grow trees on their private lands. Plantation forestry may make sense as part of a portfolio to cope with an inherently uncertain future. The current analysis abstracts from such, potentially important, considerations.

The structure of the paper is as follows. In section 2 we provide a short overview of deforestation patterns and rates in developing countries in general and in Costa Rica (and the AZ) in particular. In section 3 we distinguish different functions served by tropical forests and attach a monetary value to each of these. For this purpose we construct reasonable ranges for the study area, based on existing literature on forest valuation, ongoing research in the region, and intimate field knowledge. In section 4 we describe the opportunity cost of sustainable forest management, or agricultural profitability at the margin. In section 5 we present an optimum control model that is used to compute the socially optimal natural forest stock for the study region for various assumptions with respect to forest values and the discount rate. The final section draws conclusions and offers suggestions for extending the study.

2. Deforestation: patterns and rates

Tropical (rain)forests provide several services to humans, and it is now well documented that in many parts of the world these forests are disappearing at increasing rate. Recently, the United Nations' Food and Agricultural Organization (FAO) reported that tropical forest area has decreased from 1910.4×10^6 ha in 1980 to 1756.3×10^6 ha in 1990, which implies an annual rate of decline of 0.8% (FAO 1993). The latter has increased over time; while in the first half of the 1980s the annual deforested area was 11.3×10^6 ha, the average area deforested during this decade was 15.4×10^6 ha/year (FAO 1993). According to the FAO (1997), loss of natural forests in the first half of the 1990s was approximately 12.9×10^6 ha.[4] There appears to be a general consensus that current deforestation is one of the most pressing environmental problems of our time (Brown and Pearce 1994).

During the 1970s and 1980s, deforestation has proceeded rapidly in Costa Rica. Annual deforestation fell from between 40 000 and 60 000 ha in the late 1970s and early 1980s to 18 000 ha between 1987 and 1992, and more recently to only 8500 ha (Huising 1993; Lutz et al. 1993; Kaimowitz 1996). In the past, deforestation for low-input livestock was the dominant trend. Government policies aimed at stimulating livestock raising (and land tenure laws) were an important conducive factor for this expansion. Current deforestation is partly compensated by reforestation efforts and forest regeneration on abandoned grazing grounds (Abler et al. 1994).

The decline in the (net) deforestation rate in Costa Rica in the 1990s is no cause for comfort for conservationists. At the current rate of deforestation, all of the remaining natural

[2] Although fundamentally different forest types are present in the AZ, we do not distinguish between forest types in this study. This will be elaborated upon in future research.

[3] The model does not contain a geographical (spatial) component; that is, it does not solve for the location of the forest areas.

[4] Since most commercial logging takes the form of selective logging (i.e., where some residual tree cover remains after the bulldozers and chainsaws have moved out), the share of commercial logging in total deforestation is modest: estimates typically vary from 2 to 20% (e.g., Amelung and Diehl 1992). The general consensus is that agricultural conversion (shifting cultivation and conversion for "permanent" agriculture) is the most prominent cause. Shifting cultivation and permanent agriculture are equally damaging, with each contributing about 45% to total deforestation (Amelung and Diehl 1992).

forest will be lost in a couple of decades.[5] Even though Costa Rica has set aside 27% of its land area in national parks and reserves, nearly all forests outside these parks have been cleared. Indeed, if wood and land are considered the only valuable products contained in parks, there will exist a real possibility that these protected areas are harvested too when economic pressure increases (Tangley 1990). Because of increased land-use conflicts and more pressure on remaining forest habitats (especially by the threat of expansion of banana cultivation in the study region; see Hunter 1994), the need for an evaluation of deforestation is growing (Boza 1993). Thus, the question of proper valuing of forest functions in a monetary sense can be considered as a highly relevant issue in Costa Rica. This is especially true in view of some evidence suggesting that losses in biodiversity so far are modest (because at least some forest habitat in most ecological zones still remains) but that further deforestation may result in significant losses in biodiversity (Abler et al. 1994).

3. Tropical forest functions and their values

Tropical rainforests provide a number of services or functions. Classification of these functions is necessarily arbitrary because they are often strongly interrelated. One possible classification scheme is in terms of (*i*) production functions, (*ii*) regulatory functions, and (*iii*) habitat functions (see van Kooten et al. 1999; for an interesting analysis of a fishery combining use and nonuse values; see Costello et al. 1998). Optimal management of tropical forests requires not only that the economic value of separate functions is known but also their compatibility. In this section we will briefly summarze some relevant literature concerning valuation of the above-mentioned functions, with special reference to (the AZ of) Costa Rica.

Production functions of tropical rainforests

Tropical rainforests produce tangible products such as timber, fuelwood, and nontimber forest products (NTFP), as well as fewer tangible assets such as opportunities for ecotourism. When considering the production function of forests, the best known is the potential to (sustainably) harvest timber. Even though a standing forest can be sustainably managed, the historical performance of sustainable logging in the tropics is poor (Poore 1991). Estimates in the literature of the value of sustainable selective logging per hectare vary considerably, with differences due to (among other things) discount rates, stumpage prices, management costs, and site productivity (see, for example, Sedjo 1987; Vincent 1990; Pearce and Warford 1993; Andersen[6]).

Recently, a number of articles have been published on sustainable timber extraction in Costa Rica and the study area (e.g., Howard and Valerio 1996; Carranza et al. 1996). Several projects with the aim to sustainably manage tropical forests, are ongoing in Costa Rica (BOSCOSA, COSEFORMA, ASACODE; see Perl et al. 1991). Estimates on sustainable extraction of marketable timber from primary forest range from 0.5 to 2.0 m^3/ha per year (Carranza et al. 1996; Quiros and Finigan 1994; Solis et al. 1996, cited in Carranza et al. 1996). We use the rather conservative production estimate of 1.2 m^3/ha per year of Howard and Valerio (1996) for sustainable timber extraction in the AZ.

Timber prices used in this study were US\$50/m^3 and were arrived at by using price information from the Costa Rican Chamber of Foresters (CCF 1997) and expert knowledge.[7] In the empirical section, this price is doubled in a sensitivity analysis to allow for future changes in price. Given the size (and the growth) of the domestic market for timber, it can be safely assumed that supply in the AZ has no impact on prices.

Another source of revenues is small-scale gathering of NTFP such as rattan, oils, fruits, nuts, ornamental flowers, and bush meat; these activities are also referred to as extractivism.[8] Estimates for the economic returns of gathering NTFP span a broad range. On the high side, a frequently cited study by Peters et al. (1989) found that a particular hectare in the Peruvian rainforest yielded revenues from sustainable logging plus gathering of fruits and latex that exceeded those of timber production (clear-cutting and subsequent rotations of pulpwood). Balick and Mendelsohn (1992) calculated total annual revenues from NTFP at US\$36/ha in 30-year-old forest and US\$166/ha in 50-year-old forest for Belize. However, many authors have cautioned against extrapolating these high figures to large stretches of tropical rainforests because of, for example, downward sloping demand for NTFP and increasing costs of production and transportation.[9] In addition, Homma (1994) argues that profitable exploitation of nontimber forest products will stimulate the development of "domesticated" alternatives

[5] It is recognized that possibilities for further deforestation are limited, as most of the remaining forest is located in publicly owned protected areas and some of the land is of marginal value for agriculture. However, Veldkamp (1993) have established that even though land conversion for agriculture in the AZ has traditionally taken place preferably on fertile soils, some recent deforestation takes place on moderately fertile or infertile soils.

[6] Andersen, L.E. A cost benefit analysis of deforestation in the Brazilian Amazon. Discussion Paper, 1996. Unpublished data.

[7] Note that prices in Costa Rica are distinctly lower than prices quoted by Andersen[6] for the Amazon, who applies a price of US\$200/m^3, or prices used by Ehui and Hertel (1989) for Ivory Coast.

[8] de Beer and McDermott (1989) argue that the value of these products has been grossly underestimated for a long time. A large number of forest dwellers is believed to depend critically for their survival on this resource. The observation that forest dwellers depend on sustainable extractivism, and that this is somehow good, is open to dispute. Homma (1994), among others, criticizes the "cult of poverty" that he detects in the writings of many proponents of extractivism: they preach a return to the past and deny the problems of today, thereby promoting sustainable underdevelopment.

[9] Little is known about NTFP extraction damage to the ecosystem, or as the amount of NTFP that can be sustainably harvested, but Pearce and Warford (1993) note that exploitation of nontimber products on a large scale will probably not be compatible with the strict preservation option as "the management record [in terms of environmental damage] for many nontimber products is hardly better than that for tropical timber" (p. 121).

(viz., plantations of rubber and cocoa), which will exert additional downward pressure on prices. Also, the high value presented by Peters et al. (1989) is based on taking 75% of the inventory of the total stock of nontimber forest products, rather than on estimates of sustainable yield. Distinctly lower estimates are also available. Andersen[6], for example, arrives at US$10/ha for the Amazon. Using data from Peru, Pinedo-Vasquez et al. (1992) calculate US$20/ha per year for NTFP revenues alone, while Seción Irazábal (1996) calculates less than US$12/ha per year for timber and nontimber products together in Guatemala. Finally, Anderson (1992) reports the even lower figure of US$2.35/ha per year for the Amazon region.

As revenues from NTFP vary to a large extent in the surveyed literature, we used modest monetary NTFP values for comparable forests areas enriched by expert knowledge from Costa Rica. Unfortunately, currently no monetary estimates are available for the study area (but see Ammour et al. (1994), Barrantes et al. (1994), and Ling et al. (1996) for qualitative work). It is expected that, in the near future, some information on NTFP for natural ecosystems in the AZ will be available through ongoing research in the context of CATIE's OLAFO project (Proyecto Conservacion para el Desarrollo sostenible en America Central). We assume that gathering NTFP is able to generate US$10/ha per year when harvested on a sustainable basis, i.e., without having a major impact on other forest functions. Since supply is small compared with potential demand (for example, a considerable part of the NTFP will be exported), we assume that the demand function for NTFP is not downward sloping.[10]

There is at least one more (potential) production function that needs to be discussed, namely that of (eco)tourism. Tourism in Costa Rica became the biggest export product in 1993, with over 750 000 tourists visiting the country in 1994 (Van Leiden 1995). Based on observations in South and Central America (especially Ecuador), De Groot (1992) crudely estimates that tourism may contribute as much as US$26/ha per year to the national economy of a country with tropical forests. Ruitenbeek (1989), who estimates that the present value of tourism in Korup National Park (Cameroon) amounts to be approximately US$13/ha, provides a lower estimate. Andersen[6], using favourable assumptions concerning the development of tourist numbers in the future, produces the lower estimate of US$3/ha per year for the Brazilian Amazon. These data suggest that the role of tourism in the promotion of forest conservation is likely to be relatively small, the more so since the value of tourism in tropical forest recreation areas falls as more such area becomes available. Thus, extrapolating the findings from a small number of national parks to the global or national forest area is hazardous.

For Costa Rica, several contingent valuation studies have aimed to estimate potential tourist revenues and optimal park fees (e.g., Echevaria et al. 1995; Shultz 1997; Shultz et al. 1997). However, these studies do not quantitatively value tourism in the study region. Carranza et al. (1996) estimate that the economic benefits of tourism amount to about US$5/ha per year for primary forest and US$2.50/ha per year for secondary forest for Costa Rica. This suggests that, on average, tourism may be able to generate some US$4/ha per year. However, Costa Rica in general, and even the AZ in particular (through the exploitation of National Park Tortuguero and Refugio Vida Sylvestre Barro Colorado, as well as a few minor tourist sites), are relatively large "suppliers" of ecotourism opportunities, so it is expected that the marginal value of tourism declines as more forest areas become available. This implies that it is probably not reasonable to assume that a constant benefit per hectare for forest tourism exists. We have derived the following specification for tourism revenues (per hectare and in U.S. dollars) at the margin (MR): MR = 60 – 0.00082H, where H is the number of hectares under forest cover and accessible for recreation (see Appendix 1). This implies that the first 73 000 ha of forest in the zone are able to generate (some) positive revenues from tourism, but conserving more than 73 000 ha of forest does not generate any additional tourist revenues, and the MR is thus set at zero.[11]

Regulatory functions of the rainforest

While monetary valuation of the different production functions provided by tropical forests is, at least conceptually, a relatively straightforward matter (despite considerable uncertainties), the problem of valuing the variety of regulatory (and habitat) functions proves to be even more troublesome. This is not to say that no attempts have been made in the past. However, the degree of uncertainty that surrounds many of the estimates is distinctly greater than is the case for the production functions. For example, tropical forests are a major carbon (and methane) sink, with carbon dioxide (CO_2) the most important contributor to the greenhouse effect (Houghton et al. 1990). Houghton (1993), for example, estimates that tropical deforestation was the cause of between 22 and 26% of all greenhouse gas emissions in the 1980s. Since the benefits of greenhouse gas fixation are the discounted sum of avoided future damages, uncertainty about the extent and costs of global warming (due to changes in the frequency of extreme events, such as droughts, hurricanes, and rising sea level, and changes in agricultural productivity) has direct implications for the value of this regulatory function.

The Intergovernmental Panel on Climate Change (IPCC) and Andersen[6] provide overviews of estimates of the social costs of CO_2 emissions in different decades (also see Stavins 1999). The IPCC (1996) does not endorse any particular range of values for the marginal damages of CO_2 emissions, but published estimates of the net present value of future damage range between US$5 and US$150/t of carbon emitted (see, for example, Nordhaus 1991; Cline 1992, 1997; Fankhauser 1995; Maddison 1994), depending on, among other things, the discount rate applied to weigh future costs.

[10]It should be noted here that large-scale extraction of NTFP is not traditionally common in present-day Costa Rica, and therefore, there exits no well-developed market structure for NTFP. However, NTFP seem amply available, and different organizations are investigating their harvesting and marketing potential (Ammour 1997; T. Ammour, personal communication).

[11]For the numerical analysis we have also applied the assumption of constant benefit per hectare. The differences in terms of optimal forest stock were minimal.

The amount of carbon fixed in tropical forests as a stock depends in the first place on the standing vegetation. Navarro Monge (1996) estimated the amount of carbon stored in four different forest types in the AZ of Costa Rica at between 82.3 and 163.4 t/ha. In our model we use an average of 100 t C stored per hectare. For simplicity we assume that, for mature forests, carbon sequestration is balanced by carbon release through decay, such that net sequestration is negligible (although we allow sustainable extraction of timber and NTFP, implying some net forest growth).[12]

Costa Rica has some experience with carbon storage in the context of joint implementation (JI) programs. The OCIC (Costa Rican Office of Joint Implementation) has recently struck deals with the United States and Norway, where both parties have agreed to pay a one time payment of US$10 per metric ton of carbon fixed. This money is used to compensate private forest owners for the positive externalities associated with forest preservation.[13] Another example of JI is the CARFIX project with United States that pays carbon offset investors US$10/t carbon fixed (Foundation for the Development of the Central Volcanic Mountain Range 1993). However, like the project above, these initiatives do not extend to all the (potential) forest owners in the country (the amount of money involved is limited, and not all interested forest owners receive such compensation). Carranza et al. (1996) focus on fixation of carbon (as opposed to storage) and estimate that the net present value of benefits amounts to US$5/t. In the next section we will look at storage of carbon (or rather, avoiding emissions by conserving stored carbon), and instead of looking at net present value of damages avoided, we will vary the annual benefit per ton of carbon fixed. Annual benefit will be varied between US$0.50 (consistent with, for example, a combination of net present value of US$5 and a discount rate of 10%) and US$5 (consistent with a net present value of US$125 and a discount rate of 4%). Given the uncertainty that surrounds many of the estimates, such a broad range is justified.

Rainforests also perform several regulatory functions at the regional level, but generalizations concerning monetary valuation at this "level" are especially bothersome because the damage done when the ecological function is no longer served (or, alternatively, the cost incurred to artificially replace it after it is gone) are very much dependent on local circumstances. To take a well-known example in this context, the economic costs of protection against floods or deforestation, soil erosion, and eventual downstream sedimentation are significantly affected by the functions of the downstream area: does it have an important ecological function as a breeding ground for marine species; does it boast a rare and ecologically rich ecosystem; and does it support a profitable fishery? The loss of local functions can be considerable (see, for example, Postel and Heisse (1988), Ruitenbeek (1989)). In addition, tropical forests may regulate the microclimate, mainly via evaporation (Lean and Warrilow 1989; De Groot 1992; Fearnside 1995).

The reasoning above implies that it is impossible to draw general conclusions. Careful accounting on a case by case basis is necessary to capture the economic value of these regional regulatory functions. In the empirical analysis below we will not include the value of the regulatory functions of soil conservation, water recycling, and the watershed function. Since the study region is flat, erosion is negligible. Moreover, no reliable (economic) information is available with respect to water recycling. (See Andersen[6] for some "wild guess" work.)

Habitat functions of tropical rainforests

While the problem of valuing regulatory functions is intrinsically related to the diversity of effects and consequences, and the uncertainties that surround them, a major problem in valuing habitat functions of tropical (rain) forests is rooted in ethics. Tropical rainforests are home to not only millions of people for whom forests may be an integrated part of economic, social, and religious life but also to millions of other species, most of which are endemic to this ecosystem. However, there exists substantial uncertainty about the actual number of species that depend on tropical forests and the extent to which some of these are made extinct as a result of deforestation (see, for example, Leakey and Lewin (1995) for a discussion). Given that there is some evidence that tropical rainforests are home to more than 50% of all species in the world (Brown and Pearce 1994), this places some onus of responsibility on those whose actions result in deforestation. Does every species have an intrinsic right to life, until it vanishes from "natural causes?" If so, large-scale deforestation may be out of the question, regardless of any economic benefits that could be obtained from forest conversion. However, economists focus on anthropocentric values, hence the value of the habitat function of rainforests is determined by summing the (*i*) use values of the products and species that it supports and (*ii*) the nonuse values of biodiversity.

The use values of biodiversity have attracted the attention of economists and ecologists alike, not in the least spurred by the belief that demonstration of high values provides a convincing argument against human intervention in "vulnerable" ecosystems.[14] The rainforest may be a valuable source of new medicines. However, recent research has indicated that the use values of biodiversity may be relatively small. Simpson et al. (1996), using varying assumptions, computed "upper bounds" for the values of the marginal species, using these to derive estimates of the value of a marginal hectare

[12]The proportion of carbon in biomass varies with tree species, although it is generally in the range of 200 kg/m^3. The amount of timber harvested per hectare as used in this study (1.2 m^3/ha per year) then represents a total of 0.24 t C extracted, and thus through forest regrowth, potentially fixed annually. From this 0.24 t, only a part remains stored in the final (wood) product. We ignore this relatively unimportant figure in our final analysis.

[13]Subsidies are in the range of US$45–100/ha per year and include compensation for carbon storage, biodiversity conservation, and other regulatory functions. However, these numbers are based on negotiations and not on economic analysis (OCIC 1996*a*, 1996*b*; Tattenbach and Gorbitz, OCIC, San José, Costa Rica, personal communication).

[14]Leakey and Lewin (1995), for example, describe how lucrative and important the drugs Vincristine and Vinblastine, alkaloids from the rosy periwinkle from Madagascar, have been in curing acute lymphocytic leukemia and Hodgkin's disease.

of tropical rainforest. They value the marginal species on the basis of its incremental contribution to the probability of making a commercial discovery. Their analysis indicates that even the most favourable set of assumptions yields an estimate of pharmaceutical value of no more than US$20/ha and that for the case of an ecological "hot-spot" in Ecuador. Estimates for other regions should be much lower. The low value per hectare (and per species) is caused primarily by redundancy in genetic resources, i.e., finding a pharmaceutical compound for the second time generates little value.[15] Obviously, this marginal value will rise when the total stock (and the number of surviving species) is reduced. A similar point is made by Andersen[6] who discusses the implications of island biography (the branch of biology concerned with the relationship between species and habitat size) for the direct use value of biodiversity for the Amazon: with the current forest stock this value is believed to be less than a dollar per hectare per year, although this figure is expected to increase dramatically as deforestation proceeds.

Compared with the Amazon, the forest stock in Costa Rica is small (although experts believe that Costa Rica harbours 5–7% of the earth's biological diversity; see Tangley 1990). Therefore, it does not seem far-fetched to assume that the direct-use value of biodiversity at the margin is constant (although obviously the analysis is easily extended to allow for increasing marginal value in cumulative deforestation). The relatively low numbers for direct-use value as reported by Simpson et al. (1996) and Andersen[6] are consistent with recent work by Barbier and Aylward (1996), who conclude that "the full social costs of biodiversity protection are simply too high to be compensated through prospecting on its own" and produce an estimate of US$7/ha per year of the pharmaceutical value of biodiversity. This is roughly consistent with figures provided by Carranza et al. (1996) for Costa Rica (US$7.50/ha per year for secondary forest and US$10/ha per year for primary forests) and will be the value we employ in the empirical section below.

Obviously, limited direct use value does not necessarily imply that the economic value of biodiversity is modest. In addition to direct- and indirect-use values, nonuse values are also important. As a rough indication of the magnitude of these existence values, Pearce and Warford (1993) suggest that the existence value of tropical forests amounts to some US$8 per adult in Australia, Western Europe, and North America. This implies that total existence value is no less than US$3.2 billion a year or, dividing by the total tropical forest area (about 1750×10^6 ha in 1990), approximately US$1.80/ha. Dividing instead by the total area of tropical rainforest only (about 720×10^6 ha), existence value per hectare rises to US$4.50/ha. We assume that these values can be considered as reasonable estimates of existence value of biodiversity in Costa Rica.[16]

Biodiversity preservation is obviously more compatible with selective logging (possibly combined with small-scale extractivism) than with clear-cutting and subsequent harvesting of pulpwood or agricultural products. The degree of compatibility is open to debate (see, for example, Panayotou and Ashton 1992) and could possibly determine which land-use option is most attractive from society's point of view. In what follows we adopt the conservative assumption that the production, regulatory, and habitat functions described above are fully compatible and can be added to reflect the total value of sustainable forest management.

4. Agriculture in the Atlantic zone of Costa Rica

In the previous section we have attempted to estimate the economic value of forest functions for the AZ of Costa Rica. The rationale of forest conservation, however, depends on the opportunity cost of sustainable forest management. In the AZ, the main alternative of forestry is agricultural development. Actual and potential land use in the region have been intensively studied (e.g., Bouma et al. 1995). Land-use systems (LUS) can be defined as combinations of land units and crops (or cattle), which can be extended to so called land-use systems and technology (LUSTs) by adding a technology dimension. For example, within the study area we distinguish land units based on soil fertility and soil drainage characteristics (the climate is treated as homogenous). An example of a LUS would be pineapple on well-drained, infertile soils. Adding the technology dimension (by specifying the input–output structure in terms of yield, chemical use, labour patterns, and land preparation and maintenance methods) then defines the LUST.[17]

It is possible to (*i*) find the most profitable LUST for every land unit in the AZ and (*ii*) rank these combinations of preferred LUSTs and land units in terms of profitability (i.e., value added per hectare) from most profitable to least profitable. When highly profitable LUSTs are pursued first and

[15] "Regardless of the probability with which the discovery for the commercially useful compound may be made, if the set of organisms that may be sampled is large, the value of the marginal species must be very small" (Simpson et al. 1996).

[16] Finally, it should be recognized that land-use decisions take place under uncertainty about the values of (some) forest functions. Since tropical deforestation may well be understood as an irreversible process (at least, measured on a human time scale), the concept of quasioption value (QOV) needs to be considered. By preserving a forest, or exploiting it at moderate intensity so that its main features are retained, more options are available in future periods. Clearfelling the forest in period 0 implies that forest preservation is no longer an option in period 1 (should this be the preferred option in the light of new information), while preservation in period 0 does not rule out clearfelling at a later time. QOV thus measures the benefit of flexibility. Albers et al. (1996) demonstrate that, if two competing land uses (in our case, preservation versus clear felling and growing pulp trees or agriculture) both yield small net benefits, fairly modest QOVs can have dramatic effects on land-use choice. No data are available for Costa Rica, and therefore, QOV will be ignored in what follows (which implies that the benefits of forest conservation may be somewhat underestimated, although the results by Albers et al. (1996) indicate that these benefits are probably relatively minor).

[17] In Ehui and Hertel's (1989) study an econometric analysis was performed to analyse the link between, for example, purchased inputs and agricultural productivity. The LUST has been defined (and accordingly the revenues have been computed) based on optimal sustainable management and input application. This implies that "soil mining" is not an option and that we may underestimate the true profitability of agricultural conversion. Our data indicate that optimal sustainable agriculture in the study region earns positive rents, even on poor soils.

less attractive options later, then expanding the agricultural frontier implies moving to less profitable options. This allows an evaluation of the profitability of agriculture at the margin.

The specification of the function that describes agricultural productivity is as follows. From the exercise above we have a single observation for each distinguished class of land units. For example, growing banana on well-drained, fertile soils yields a value added of approximately US$3000/ha per year, while raising cattle on poorly drained, infertile soils yields no more than US$125/ha per year. We have fitted a curve through these observations (with annual revenues in U.S. dollars on the vertical axis and number of hectares on the horizontal axis). The marginal revenues associated with different land units can be imagined as bars of a histogram, where the height of a bar indicates profitability and the width is a measure of the number of hectares in this particular class of land units. The curve is fitted through central values of each bar. Hence, for the banana case above, when 100 000 ha classify as well-drained and fertile, then the marginal revenue curve is fitted through the observation (50 000 ha, $3000).

We have fitted several possible specifications for the function that describes marginal productivity at the margin (linear, quadratic, and exponential functions). In the next section we will present results for the linear specification: $A = 3459 - 0.006x$, where A is marginal agricultural revenues and x is the number of hectares in agricultural production ($R^2 = 0.91$). For example, when $x = 300\,000$, converting an additional hectare of forest in agricultural land will yield an additional annual revenue of approximately US$1660. The results are fairly robust with respect to the specification of A.[18] Agricultural profitability at the margin for the latter specification is depicted in Fig. 1, where A is on the vertical axis and x is on the horizontal axis.

We assume that the region faces perfectly elastic supply and demand curves for inputs and outputs. As the region is relatively small (and only part of it will be allocated to any specific crop due to different soil requirements), this assumption seems innocuous. The assumption of a horizontal supply curve of labour may perhaps appear strong, but it should be realised that there is ample (potential) labour available in the region due to immigration of labourers from Nicaragua.

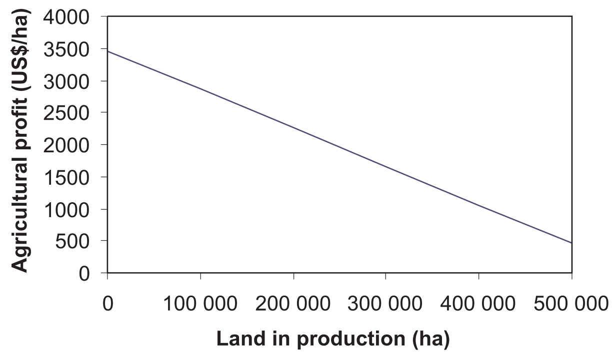
5. Determining optimal forest stocks in the Atlantic Zone of Costa Rica

The formal optimal control model (suppressing time notation) is as follows:

$$(1) \qquad \text{Max } W = \int_0^\infty \pi(t)\mathrm{e}^{-rt}\,\mathrm{d}t$$

where

Fig. 1. Marginal agricultural products as a function of land in production.



$$(2) \qquad \pi = B[F(t)] + \int_0^{S-F} A(z)\,\mathrm{d}z + \tau D(t)$$

subject to:

$$(3) \qquad \dot{F}(t) = -D(t)$$

where $\pi(t)$ is economic benefits; $B(F)$ are benefits of sustainable forest exploitation; $F(t)$ is the forest stock at time t; S represents the initial forest stock, equal to the entire AZ region, so that $S - F$ is land devoted to agricultural production or x); τ represents one-time timber benefits of converting a hectare of forest land to agriculture (see van Leeuwen and Hofstede 1995); and $D(t)$ is the deforested area at time t. Note that the equation of motion (eq. 3) captures only deforestation and not the possibility of reforestation or plantation forestry. This is due to the fact that the focus of the paper is on conservation of natural forests. It is assumed that natural and plantation forests are imperfect substitutes for most of the forest functions discussed in section 3 (exceptions include the timber production function and the carbon sequestration function), so that deforestation can be considered an irreversible process. The term $\int_0^{S-F} A(z)\,\mathrm{d}z$ describes total benefits of agriculture for the $S - F$ hectares of agricultural land. $B(F)$ is defined as the sum of benefits from sustainable timber and NTFP exploitation, ecotourism, carbon fixing, and use- and nonuse values of biodiversity conservation.[19] Finally, r is the social discount rate. Maximization takes place subject to the equation of motion provided in eq. (3), where the dot over the variable F indicates a time derivative.

The current-value Hamiltonian, while omitting time notation, is defined as follows: $H = \pi - \lambda D$, where λ is the costate variable, akin to a Lagrange multiplier. Assuming an interior solution, the necessary conditions for an optimum solution are

$$(4) \qquad \frac{\partial H}{\partial D} = 0 \rightarrow \lambda = \tau$$

[18] We have tried an exponential specification, and the resulting land allocation was roughly the same. We do not report the results of sensitivity analysis.

[19] Forest benefits are defined as $B(F)$ because benefits from tourism at the margin are a function of F (see Appendix 1).

Table 1. Socially optimal forest stocks (F^*) in hectares for different values of carbon fixation (C), stumpage price per cubic metre (P), and discount rate (r).

	C = 0.5		C = 2		C = 3.5		C = 5	
r (%)	P = 50	P = 100	P = 50	P = 100	P = 50	P = 100	P = 50	P = 100
0	0	1200	16 200	26 200	41 200	51 200	66 200	76 200
3	0	0	11 200	16 200	36 200	41 200	61 200	66 200
6	0	0	6200	6200	31 200	31 200	56 200	56 200
9	0	0	1200	0	26 200	21 200	51 200	46 200
12	0	0	0	0	21 200	11 200	46 200	36 200
15	0	0	0	0	16 200	1200	41 200	26 200

$$\dot{\lambda} = r\lambda - \frac{\partial H}{\partial F} \rightarrow \dot{\lambda} = r\lambda - [B'(F) - A(S - F)] \quad (5)$$

The interpretation of eq. 4 is that the rate of deforestation should be chosen such that the marginal utility of current deforestation τ is equal to the opportunity cost of harvesting the value of having access to the stock in the future λ. Equation 5 provides a standard intertemporal nonarbitrage condition (Ehui and Hertel 1989; Clark 1990).

The steady state occurs when the costate multiplier and the forest area are constant ($\lambda = F = 0$), so no further deforestation takes place ($D = 0$). The equation that describes the optimal forest stock in the steady state is

$$\tau + \frac{A(S - F^*)}{r} = \frac{B'(F^*)}{r} \quad (6)$$

Equation 6 says that, in equilibrium, the present value of benefits of sustainable forestry management at the margin must be equal to the sum of immediate benefits of forest conversion and the present value of subsequent agricultural production at the margin. As the Hamiltonian is linear in the control variable D, optimal approach is of the bang–bang type (a most-rapid approach path).

To determine optimal forest stocks (F^*), the above model was solved numerically for the values of the forest functions and the agricultural benefits curve described in sections above. The annualized benefits of carbon fixing (C),[20] the stumpage price per cubic metre (P), and the discount rate (r) were all parametrically varied (Table 1). Assumptions with respect to the benefits of C and r significantly influence the optimal forest stock in the steady state. Optimal forest stock increases with C for obvious reasons and declines in r as future benefits of carbon-fixing, sustainable forest management and agriculture receive less weight (such that the immediate benefits of deforestation become relatively more important). Also, note from Table 1 that the relation between F^* and stumpage price P is ambiguous. This is because higher prices increase both the benefits of sustainable forest management and the benefits of current forest conversion. The relative weight again depends on the discount rate applied.[21]

For positive and zero discount rates, F^* is (substantially) lower than the current forest stock of approximately 150 000 ha.[22] This implies that further deforestation may be warranted when social welfare maximization is the objective of policy makers. This is true even when nonuse values and transboundary benefits of sustainable forest management are taken into account. Since the bulk of Costa Rican forests are located in protected areas, we conclude that the government of Costa Rica may have set aside too much (as opposed to not enough, as claimed by some critics) of its forests. This is especially true when we realise that in natural parks selective logging is not allowed, such that F^* in Table 1 may be an overestimation of the "true" optimal stock.

In the analysis we have assumed that policy makers in Costa Rica take the benefits of carbon fixation fully into account. Currently, most governments in developing countries are not fully compensated for the carbon that is fixed, even though the government of Costa Rica receives some compensation, as described above. This implies that the figures in the upper left-hand corner of Table 1 may be most relevant for policy makers in developing countries. Large-scale deforestation (possibly up to the point of total stock depletion) is optimal when the benefits of carbon fixation are not taken into account. Developed countries should therefore take international compensation for carbon fixation seriously if they aim to promote conservation of tropical forests. The opportunity cost of sustainable forest management may be considerable (as is the case in the AZ of Costa Rica), and it is an open question whether developing countries are willing and able to pay for this cost alone.

In Table 2 we have increased the sum of the value of NTFP collection, biodiversity prospecting, and nonuse values from about US$22/ha per year to US$75/ha per year. This is interpreted as an upper bound for these benefits associated with sustainable forest use. As expected, considering

[20]Obviously, the annualized benefits of carbon fixing are a function of the discount rate itself. Because of lack of data we have resorted to the approach in Tables 1 and 2, where various assumptions with respect to annual benefits (based on the range found in the literature) are arbitrarily combined with a reasonable range of values for the "social discount rate." This way we apply a broad range of estimates for discounted annual benefits of carbon fixing.

[21]Certification of sustainably produced timber presumably results in a "gap" (the green premium) between the timber price of sustainably produced wood and the timber from forest conversion. This implies that the ambiguity disappears, and certification thus unambiguously promotes forest conversion.

[22]It is possible to increase C further so that F^* is actually greater than the current stock. However, this would imply using carbon fixing benefits in excess of upper bounds found in the literature.

Table 2. Socially optimal forest stocks in hectares for different values of carbon fixation (C), stumpage price per cubic metre (P), and discount rate (r) with NTFP = US$75/ha per year.

	C = 0.5		C = 2		C = 3.5		C = 5	
r (%)	P = 50	P = 100	P = 50	P = 100	P = 50	P = 100	P = 50	P = 100
0	1200	11 200	26 200	36 200	51 200	61 200	76 200	86 200
3	0	1200	21 200	26 200	46 200	51 200	71 200	76 200
6	0	0	16 200	16 200	41 200	41 200	66 200	66 200
9	0	0	11 200	6200	36 200	31 200	61 200	56 200
12	0	0	6200	0	31 200	21 200	56 200	46 200
15	0	0	1200	0	16 200	11 200	51 200	36 200

this upper bound unambiguously increases F^*. However, since for all combinations of parameters the optimal forest stock size remains below the current stock, we conclude that our results are robust.

As a final sensitivity analysis we have explored the impact of increasing the quantity of timber that can be sustainably produced from 1 ha of natural forestland. Increasing the annual timber flow from 1.2 m^3/ha to the maximum value in the literature of 2 m^3 obviously results in a larger optimal forest stock, as it increases the profitability of forest conservation relative to agriculture. For example, consider the combination of parameters p = 50, C = 2, and r = 0.06. Increasing the forest's productivity implies that the optimal forest stock approximately doubles to almost 13 000 ha. Similarly, for the combination p = 100, C = 3.5, and r = 0.09, the optimal stock increases by somewhat less than 50% (to about 34 000 ha). However, the optimal forest stock never exceeds 100 000 ha and is thus consistently smaller than the current stock.

6. Discussion and conclusions

In this paper we have computed socially optimal forest stocks for the Atlantic Zone of Costa Rica, where both (*i*) use and nonuse values, and (*ii*) domestic and transboundary services are taken into account. The results provide a benchmark to evaluate past and current deforestation. It turns out that optimal forest stocks as computed by our model are consistently lower than current forest stocks. Only if there is good reason to believe that benefits of carbon fixing are extremely high (i.e., exceeding upper bounds currently found in the literature and in practice) and if the government in Costa Rica receives full compensation from the international community for the carbon that is fixed, can the current forest stock considered to be about optimal or maybe even too small. We have assumed that demand for agricultural products and supply of production factors in the region is perfectly elastic (which may be consistent with real life because the study region is less than 550 000 ha). Nevertheless, the results strongly suggest that the government of Costa Rica may have set aside more of its natural forests than is socially optimal. Therefore, developing part of the natural heritage in the future that is currently protected by law should receive serious consideration from policy makers (see Bromley 1999 for some interesting reflections on this matter).

We would like to discuss three important caveats to this conclusion. First, a shortcoming of the paper is the lack of a spatial dimension. Adding such a dimension would greatly affect the complexity of the model but would allow incorporating land heterogeneity in the function describing the benefits of forest conservation. Land quality is likely to affect both agricultural rents and the value of forest functions (e.g., land quality is likely to determine the type of forest that it supports, and thus is a major determinant of, for example, carbon sequestration). Having a land quality gradient for both the agricultural rent function and the function describing the benefits of forest conservation would enable detailed matching of the relative profitability of forest conservation on a per-hectare basis, rather than considering the aggregate quantity of forest to conserve. Thus, it would produce more reliable outcomes than the current project, where we simply consider estimates of forest benefits for average soil quality. The spatial component, however, would require more detailed data, which are currently not available.[23]

Another caveat that deserves some attention is the fact that the value of a hectare of Costa Rican forest increases only slightly as it the stock is drawn down. (In the current analysis, the marginal value of forest increases slightly after the stock is smaller than 73 000 ha, which is a result of the assumption of declining marginal tourism values of forest.) Most of the benefits associated with forest conservation are transboundary, (e.g., nonuse values, biodiversity prospecting, carbon sequestration), and for people in, say, Europe it makes little difference whether natural forests are conserved in the Amazon or in Costa Rica. This implies that we can assume there are ample substitution possibilities from a global perspective, and that the marginal value of forest conservation in the study region is approximately constant. It could be argued, however, that forests in different regions are

[23]An alternative approach to a detailed comparison of the profitability of forest conversion and conservation on a per-hectare basis is to divide the study region in four subregions, based on land quality. For each subregion it is then possible to proceed along the lines set out in this paper and determine the optimal forest stock. Aggregating over these four subregions then results in an improved estimate of the optimal forest stock for the entire study region. This is left for future work. Currently, we do not have access to forest data that make it useful to distinguish between different regions (other than differences in carbon stored per hectare).

imperfect substitutes for some forest functions (e.g., nonuse values). Accommodating this concern could potentially change the dismal picture painted in this paper. For example, Bulte and van Kooten (1999) have explored the consequences of declining marginal preservation value for resource conservation, and found that the impact was quite significant.[24] Assuming that the average preservation value for forest conservation amounts to $4.50/ha but that the value at the margin is a linear downward sloping curve MPV = $K - aF$ (where K and a are constants; see Bulte and van Kooten 1999), we have repeated the empirical analysis in section 5. It turns out that the changes in the optimal forest stock are extremely small, which is perhaps to be expected given that nonuse values are a relatively minor component of forest conservation (unlike the case of whale conservation considered by Bulte and van Kooten 1999). We therefore conclude that the numerical results are robust with respect to declining marginal preservation values. However, this does not imply that the scarcity value of the forest should be considered as constant. Rather, it is conditional on the global forest stock, and if deforestation in other regions of the world proceeds, the value of forest in the study region should increase.

As a final caveat it should be emphasized that ecological services are valued at zero in the empirical analysis, implying that, for example, we assume that there are no linkages between forest services and agricultural production. This is a strong assumption, at odds with findings of Ehui and Hertel (1989, 1992). Especially when forest stocks are completely depleted, as prescribed in some of our scenarios, it can be expected that some negative feedback occurs. (Note that our estimate of agricultural rents is based on the assumption of sustainable agriculture. It may well be the case that external input use should increase after the natural environment has been altered significantly and, for example, water quality has worsened). The omission of ecological services implies that we underestimate the true forest value and optimal forest stock. The lack of information on (some) public values of forest conservation combined with the strong private signals to convert forest areas will result in excessive deforestation; therefore, we conclude that future research in this field is very important.

We would like to emphasize that we are likely to underestimate the total tree cover in the study region as we have excluded plantation forestry as a possible land use option. While, as stated, plantation forests and natural forests are imperfect substitutes for most of the forest functions we have distinguished, plantations produce timber and sequester carbon. Ignoring plantation forests as a land-use option produces biased results when timber and carbon are the main concern of the government. Preliminary data for the study region indicate that planting trees (especially teak and melina species) may be privately profitable. Focussing on timber yields, Marciano (1999) estimates that up to 30% of the study region may be allocated to plantation forestry in the future. Including carbon credits (or raising timber prices) will increase this share. Since plantation forests and natural forests will have to “compete” for land (as do natural forests and agriculture in the current model), it is expected that the area of natural forest will be reduced to a greater extent than estimated in this study. To fully explore this issue, LUSTs for plantation forestry are developed and included in the land use model in section 5. We conclude that some forest functions (specifically carbon sequestration and timber production) will be supplied more than can be expected on the basis of our study, partly at the expense of the remaining functions.

In any event, the analysis illustrates a major problem with natural forest conservation in developing countries: the opportunity costs may be considerable. However, the observation that more than our estimate of the socially optimal forest stock is currently being preserved in parks may indicate that there exist wider motives for forest conservation than economic incentives alone. These motivations may be rooted in ethics. As yet it is an open question whether preservation based on this sentiment will prove to be sustainable or not.

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[24] To be specific, for the case of whaling they argue that both strict preservation and a complete moratorium on harvesting can be advocated, depending on the assumption with respect to how fast marginal preservation value falls if stock size increases (keeping total preservation value constant).

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Appendix 1

If we take the estimate of US$4/ha per year as a benchmark and assume that marginal benefits of tourism are a linear, downward sloping function ($\mathrm{MR} = \psi - \alpha F$), then it is possible to solve for this function by using additional information with respect to the tourist "hot spot" of the region: Tortuguero National Park. The entire AZ consists of about 545 000 ha; hence, we assume that total potential benefits for the zone equal roughly US$2.18 million/year. Total benefits are equal to the area under the (assumed linear) curve that depicts marginal benefits of tourism. According to Van den Broek and Van Bentveld (1994), Tortuguero National Park generates approximately US$60/ha per year. When we take this value as the point where the marginal benefit curve crosses the ordinate (i.e., assume that $\psi = 60$), it is possible to estimate the slope of the marginal benefits curve α. Define Z as the point where the marginal benefit of forest tourism equals 0.5. Then, $0.5\,\psi Z = 2\,180\,000$, so that $Z = 73\,000$ (hence $\alpha = 0.00082$).