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A dynamic model of patterns of deforestation and their effect on the ability of the Brazilian Amazonia to provide ecosystem services

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Abstract

This paper presents a dynamic systems model that shows how different land use patterns degrade the value of ecosystem services provided by the Brazilian Amazonia. The model consists of four sectors: (1) deforestation drivers; (2) land use/cover; (3) ecosystem services; and (4) ecosystem valuation. The deforestation drivers sector models the economic and social incentives that small farmers and large pasture investors have for clearing the forest. The land use/cover sector shows how these different groups clear land, and further shows how patterns of forest succession and associated biomass differ by primary land use type. Different land use patterns greatly impact the quality and economic value of ecosystem services. These impacts are dealt with in the ecosystem services sector, which models the region's hydrological cycle, the nutrient cycle, carbon sequestration capacity, and species diversity. Calculations are made in the ecosystem valuation sector according to a reference monetary value for these ecosystem services. The model calculates the change in these values according to the land use practices that occur over time. Findings show that over a 100-year simulation, forest area remains about 44% of original area with pasture and abandoned pasture becoming the dominant land cover. The value of ecosystem services declines from \$1431 to \$658 and \$781 ha^{-1} $year^{-1}$ for agriculture and pasture, respectively. These findings are compared to annual revenue derived from different land use practices for which land was cleared in the Brazilian Amazonia. In the context of these findings, the authors discuss how an explicit monetary valuation of ecosystem services could create positive incentives for land stewardship and conservation. © 2001 Elsevier Science B.V. All rights reserved.

Keywords: Deforestation; Brazilian Amazonia; Tropical forest

1. Introduction

Massive deforestation in Brazilian Amazonia, the largest continuous region of tropical forest in the world, is known to have profound effects on the forest's biological diversity, resilience to disturbance, soil and water resources, and regional

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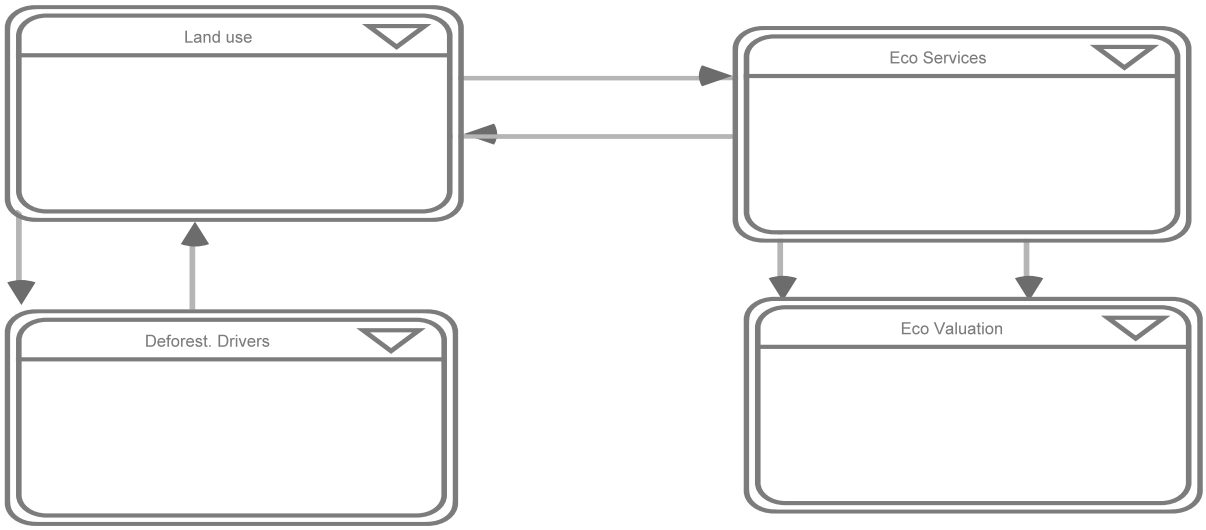

Fig. 1. Model overview.

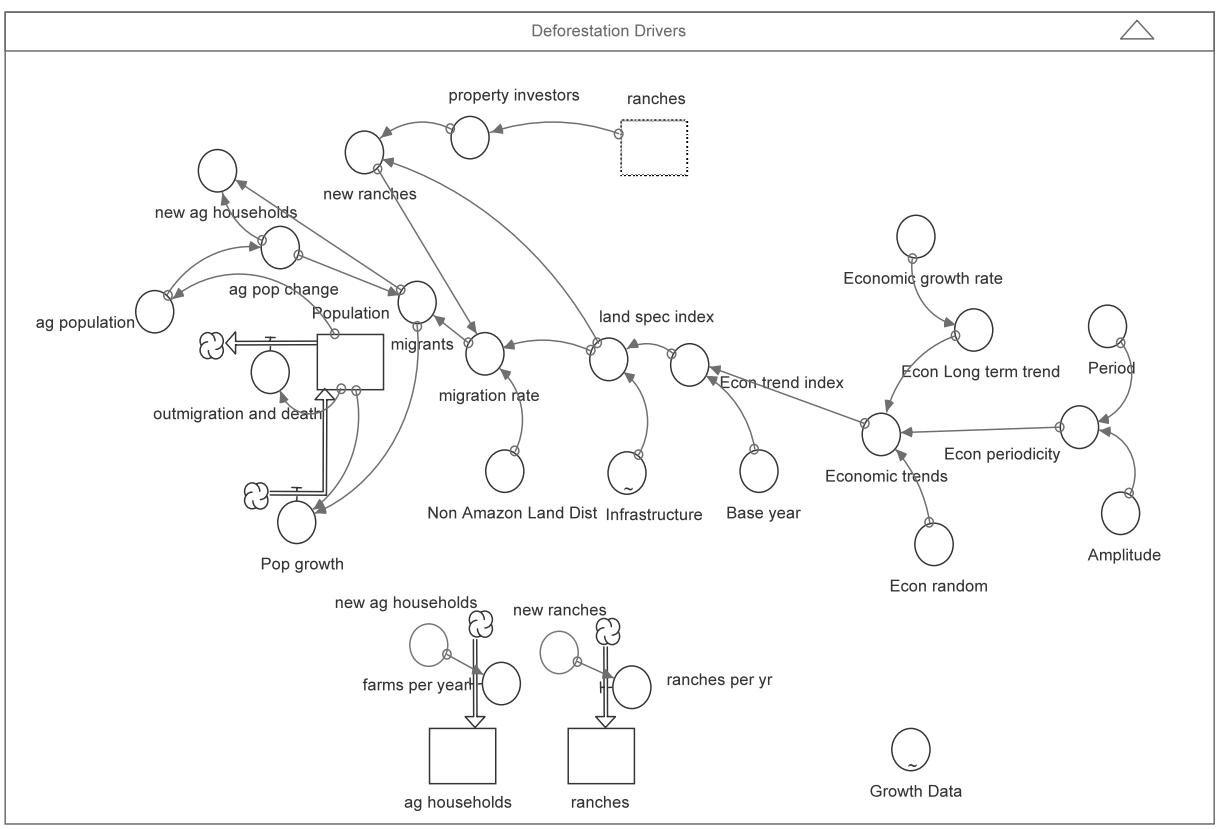

Fig. 2. Deforestation drivers sector.

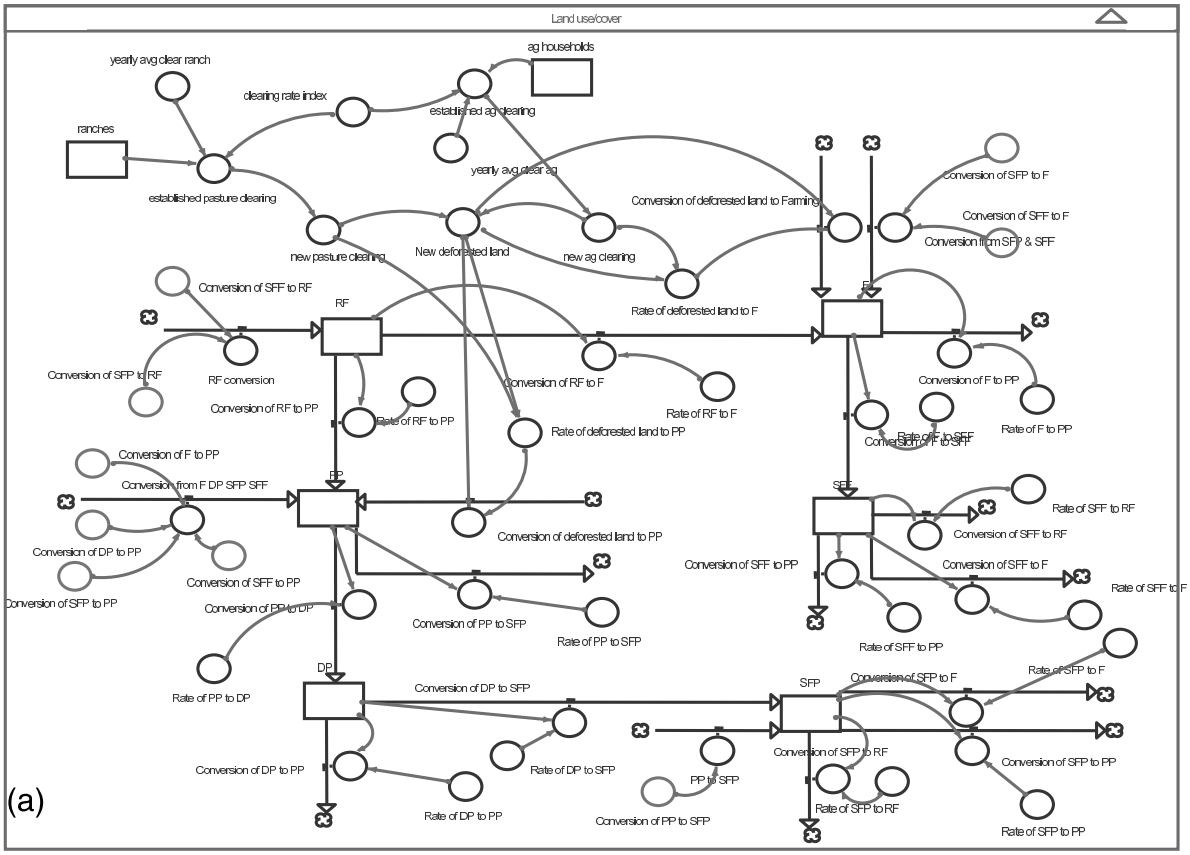

Fig. 3. (a) Land use/cover sector: transition rates. (b) Land use/cover sector: biomass amounts.

and global climate patterns (Salati and Vose, 1984; Salati, 1987; Crutzen and Andreae, 1990; Shukla et al., 1990; Salati and Nobre, 1991; Dale et al., 1993; Skole and Tucker, 1993; Dale et al., 1994; Rocha et al., 1996; Serrao et al., 1996; Wood and Perz, 1996; Zhang and Henderson-Sellers, 1996; Zhang et al., 1996; Fearnside, 1997b). The economic benefits derived from deforestation of Amazonia come from extractive, productive, and speculative practices that are encouraged by the increasing infrastructural development of the region (Hecht, 1985). Some of the main activities include logging, mining, cattle raising, agriculture, construction of dams, roads, and urban settlements (Hall, 1986; Serrao et al., 1996). The pattern of forest exploitation is based on the utilization of resources with very little or no attention paid to the value of protected forests in providing ecological functions such as biodiversity maintenance, carbon storage, nutrients cycling and erosion control (Fearnside, 1997a). The neglect of these goods and services is not puzzling given that most individuals who exploit the resources of the Amazon do so for monetary gain and nature's services are primarily non-market—and hence non-priced-goods (Faminow, 1998). The neglect is distressing, however, especially in the case of vital life support functions such as gas and climate regulation. These ecosystem services have been tremendously affected by the last few decades of clearing (Fearnside, 1996, 1997b).

In the model discussed here the authors focus primarily on deforestation that is driven by productive and speculative purposes. The purpose of the model is to understand at a very broad and aggregated level the toll that these patterns have exacted on the ecological functions and ecosystem services provided by Brazilian Amazonia. In an effort to find meaningful ways to discuss the

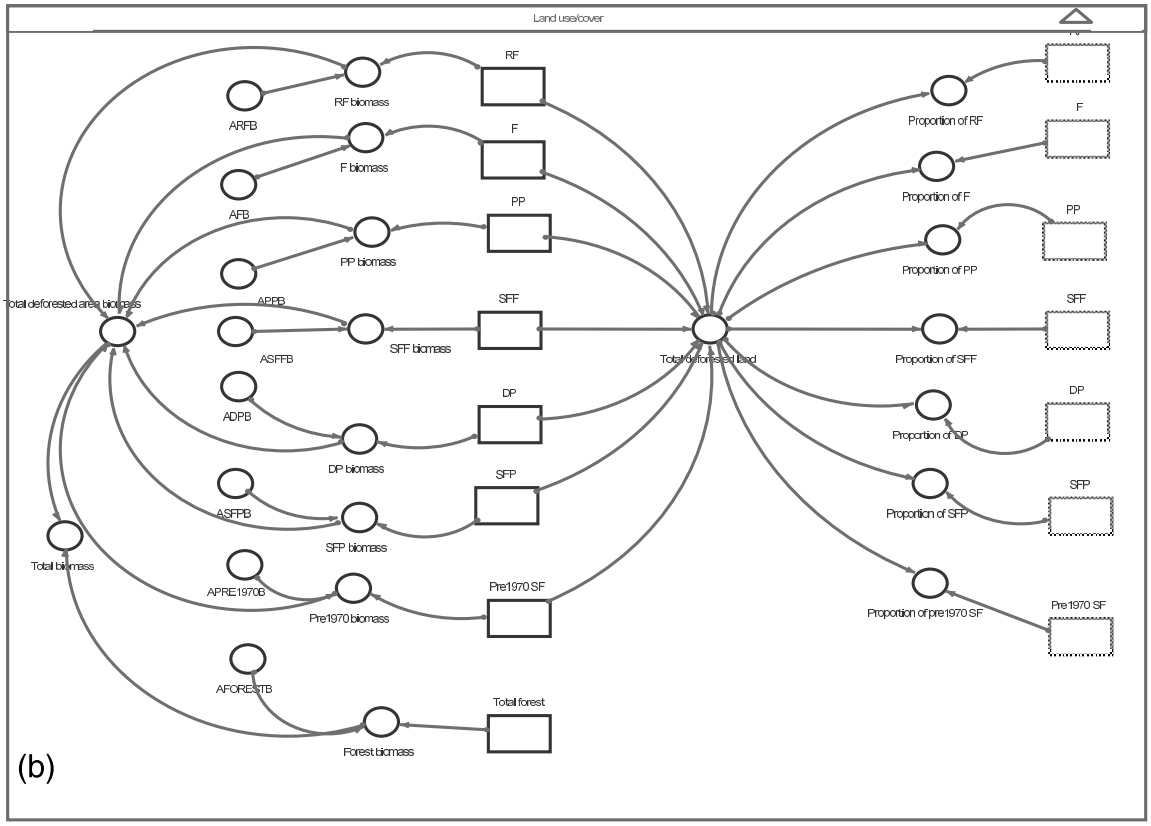

Fig. 3. (*Continued*)

importance of ecosystem services for sustained economic activity, the loss of services observed in the model due to ranching and farming land use practices is translated into an annual monetary value that can be compared to the annual revenue generated by ranching and farming activities.

1.1. Study area

The area of study consists of the Brazilian Amazonia's river drainage basin, an area of approximately four million square kilometers, encompassing the states of Acre, Amapa, Amazonas, Maranhao, Mato Grosso, Para, Rondonia, Roraima and Tocantins. Historically, this area has been primarily forested, but this is changing as land is submitted to an intense process of deforestation that started in the 1970s as a result of governmental policies designed to settle the region and exploit its natural resources. Government investment in road construction and new settlements began on a massive scale in order to alleviate population pressure in the Brazilian Northeast, strengthen Amazonian borders and enable access to the region's vast supply of resources (Pyne et al., 1996). Government-financed pro-

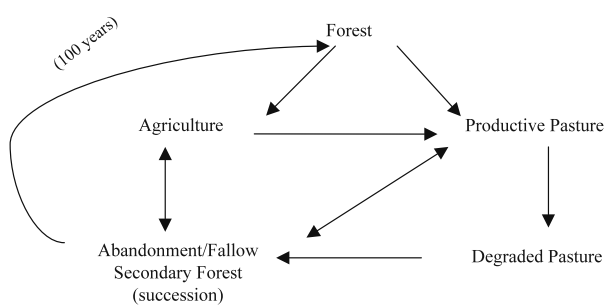

Fig. 4. Land use transition patterns from Fearuside (1996).

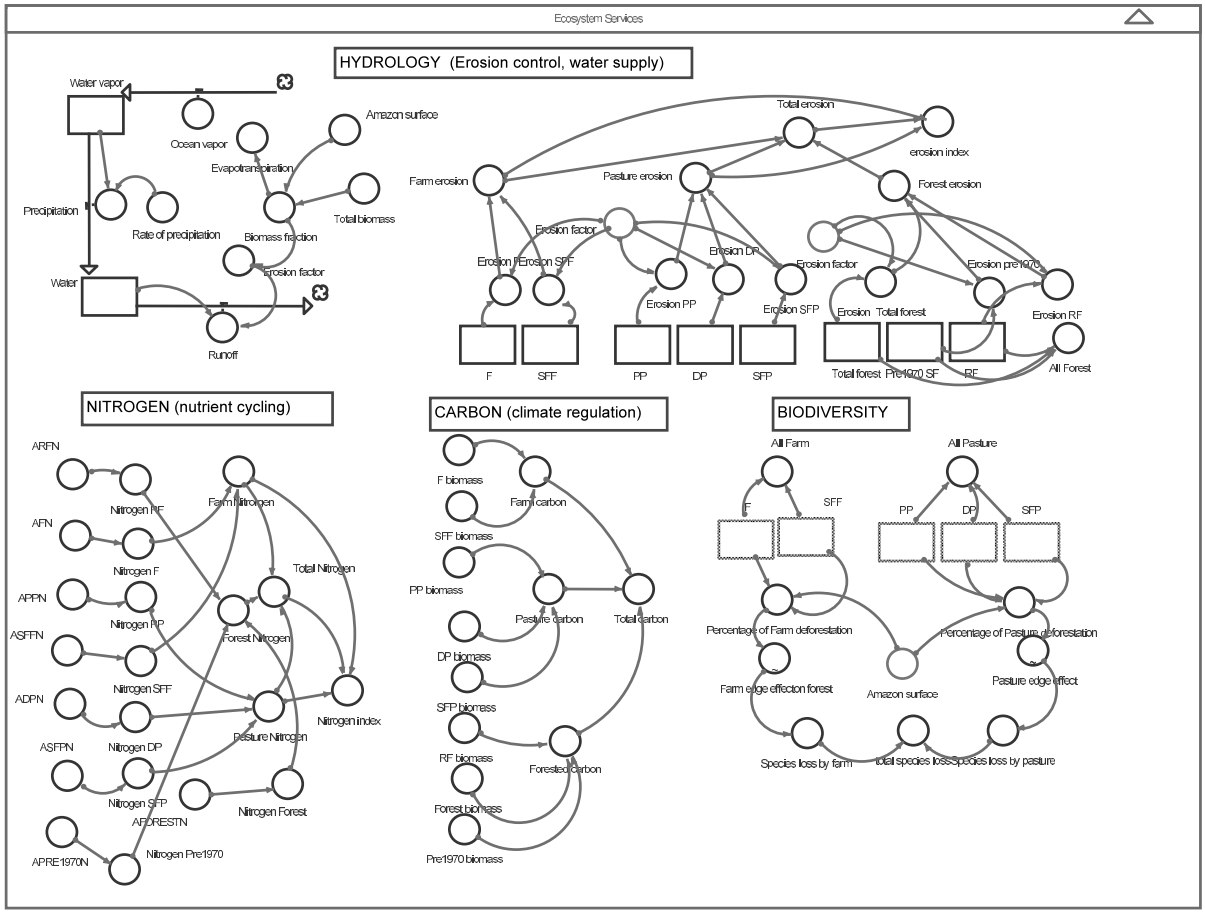

Fig. 5. Ecosystem services sector.

grams and subsidies encouraged extensive cattle ranching, farming and logging (Moran, 1991; Moran et al., 1994; Laurance et al., 1998b) along the newly created roads, especially on the southern and eastern fringes of the basin, where vast areas have been cleared and converted to pasture (Uhl et al., 1988). According to Eden et al. (1990), Fearnside (1996, 1997b), Neil et al. (1997), Uhl et al. (1988), and Walker et al. (2000) cattle ranching activities continue to account for most of the deforestation in the Brazilian Amazonia.

Recent data on the extent of this deforestation shows that about 13% of the Brazilian Amazonia Forest has already been cleared, and that the annual rate of deforestation in the last 20 years varied between 0.30 and 0.81% (INPE, 1998). In terms of area, this is equivalent to low values such as the 13 020 km^2 deforestation occurring in 1978 to the high rates of 29 160 km^2 in 1986 (ibid.). At the current rate of deforestation and with large areas yet to be cleared, an increase in the severity of the ecological and climate effects is expected (Fearnside, 1997b).

2. Methods

The model uses the STELLA programming language (High Performance Systems 1993) to explore the farming and ranching uses of the Brazilian Amazonia and the effects these practices have on ecosystem services and functions. The

Table 1
Average ecological values and associated land use categories

Land use stock	Average above ground biomass (MT ha^{-1})	Average carbon content of biomass (MT ha^{-1})	Average nitrogen in top 25 cm of soil (MT ha^{-1})	Average soil loss-erosion (MT ha^{-1})
Forest	272[a]	122.00	7.0[g]	116[h]
Pasture	10[b]	4.50	4.5[g]	580[i]
Degraded pasture	3[c]	1.35	4.0[g]	812[i]
Secondary forest from pasture	17[d]	7.65	6.0[g]	348[i]
Farm	1[e]	0.45	3.5[g]	464[i]
Secondary forest from farm	29[f]	13.05	6.0[g]	290[i]

[a] From Fearnside (1992a).
[b] From Olson et al. (1983).
[c] From Fearnside and Guimarães (1996).
[d] From Uhl et al. (1988).
[e] From Fearnside and Guimarães (1996)
[f] From Fearnside and Guimarães (1996)
[g] Estimated by the authors based on Brown and Lugo (1990).
[h] From Salati and Vose (1984).
[i] Estimated by the authors based on by Salati and Vose (1984) and Lavelle (1987).

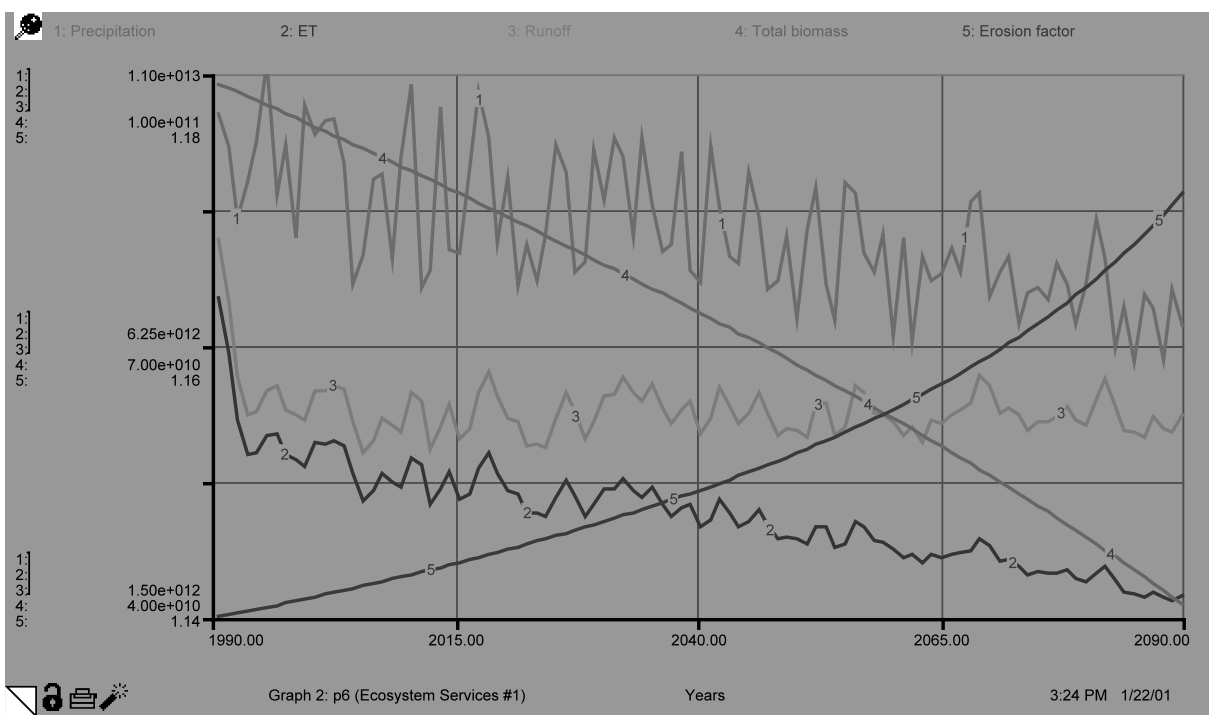

Fig. 6. Hydrology and erosion processes.

Table 2
Value of ecosystem services

Ecosystem services	Forest reference[a] ($ ha year^{-1})	Farm ($ ha year^{-1})	Pasture ($ ha year^{-1})	Total Amazon ($ year 1E6^{-1})
Climate regulation	223.00	7.00	11.00	33 972.00
Erosion control	245.00	66.00	61.00	50 849.00
Nutrient cycling	922.00	556.00	677.00	303 397.00
Genetic resources	41.00	29.00	32.00	14 048.00
Total	1431.00	658.00	781.00	402 266.00

[a] From Costanza et al. (1997).

model is highly aggregated and construction involved the elaboration of non-spatially explicit socioeconomic and ecological processes and patterns. The resulting dynamic model contains important linkages and feedbacks between human activities and ecological impacts. Links and relations between and within the different sectors of the model were developed by establishing direct and indirect connectors between state and auxiliary variables. Equations and random numbers were employed to describe some expected behaviors that are well documented in literature. Data used in the calibration of the different processes were collected from many publications and were important in checking the behavior of different sectors of the model. While some parts of the model such as ecosystem service values could not be calibrated using published quantitative data (because none could be found in the literature), there was a substantial amount of qualitative data that was used to inform all aspects of model construction. Although each aspect of the model was carefully researched, it is important to keep in mind that all results are experimental and highly aggregated. They are offered merely as a starting point for further discussion and research aimed at finding useful ways to describe the damage that is done when public goods are not valued privately in decision-making processes.

3. Model description

A dynamic simulation model was developed in order to investigate the effect of different patterns of deforestation of Brazilian Amazonia on the area's ability to provide ecosystem services. Fig. 1 shows a model overview with its major sectors. In the model, deforestation is driven by socioeconomic processes laid out in the deforestation drivers sector. Smallholders and ranchers have multiple economic and social incentives to clear land. Economic gain comes from both productive and non-productive (i.e. speculative) uses of land, and can vary due to fluctuations in the economy, ability to gain clear title to land, and access to markets.

While some incentives to clear are similar for farmers and ranchers, the patterns of clearing and land-use intensity are markedly different. These differences are reflected in the land-use/cover sector. This part of the model deals with transition rates between productive farm and pasture, degraded pasture, and secondary regrowth. The different land use patterns and land-cover change processes alter a variety of ecosystem properties, the most important of which is vegetation cover (biomass).

The effects of biomass changes are modeled in the ecosystem services sector, which includes hy-

Table 3
Total deforestation by land use category

Land use stock	Total area (km^2)	% of total deforested land
Farm	132 176	5
Secondary forest from farm	53 196	2
Productive pasture	1 215 147	46
Degraded pasture	113 428	4
Secondary forest from pasture	1 045 139	40

drology, erosion, nutrient cycling, carbon storage, and species diversity processes. The ecosystem valuation sector relates the changes that occur in the ecosystem services sector to monetary values using the values calculated by Costanza et al. in the 1997 publication 'The value of the world's ecosystem services and natural capital'.

Graphs of model behavior and tables displaying model results are presented using the user interface capabilities of STELLA. Using this interface, one can alter many of the assumptions that are used to construct the model in order to build different scenarios about deforestation, transition rates and economic activity.

3.1. Deforestation drivers sector

This model sector is shown in Fig. 2 and it depicts basic social, demographic and economic processes that have been researched and found to be significant factors in Brazilian Amazonia deforestation (Fearnside, 1987, 1993; Moran, 1991; Hecht, 1993; Mahar and Schneider, 1993; Monbiot, 1993; Wood and Perz, 1996; Pfaff, 1999). The purpose of including these factors in the model is to show in an explicit way that deforestation is largely the result of a socioeconomic process (Dale et al., 1993). By including the human dimension of deforestation in the model—even in a simplified and stylized context—it is possible to communicate something about the linkages that exist between socioeconomic and ecosystem processes, and to begin to explicitly identify the losses in ecosystem services that are directly attributable to certain types of economic activity.

The model accounts for clearing by new farm and ranch start-ups as well as clearing by existing establishments. Two main processes combine to determine deforestation by new Amazonia farms and ranches–economic incentives and population growth. Economic incentives include economic trends and infrastructure development. An economic trends index was designed to roughly mirror the highly fluctuating Gross National Product of the Brazilian economy and was calibrated to existing GNP data for Brazil. The economic trends consist of an economic long term trend that assumes growth over the long run for the Brazilian economy, and an economic short term trend that assumes that there will periods of economic expansion and recession over shorter time spans (15 years). Infrastructure refers to the density of roads in the Amazon. The infrastructure element of the submodel represents 'infrastructure density' in Amazonia in graphical form. The relationship between the economic trends index and the infrastructure density is multiplicative and the two factors form a 'land speculation index'. In this model, the index is intended to reflect the way that different incentives for development compound one another and influence the rates at which land speculation and clearing take place. The compounding factors of easier access and economic growth increase incentives for both ranch investment and migration to the Amazon.

The model assumes that the reasons for new ranch and new farm clearing are somewhat different. New Ranches are primarily a function of speculative investment, whereas new farm clearing is much more closely tied to factors such as the shifting nature of cultivation and political and economic conditions that drive population influx into the Brazilian Amazon (Hecht, 1993). Population growth includes existing settlers and new migrants (migration is further influenced by land shortages elsewhere in Brazil—represented in the model as Non Amazon land distribution). The average amount of land cleared by new farms is initially set at 3 ha (Fearnside, 1993). The average new ranch is set to clear 50 ha (Fearnside, 1993). These clearing rates can be altered in the user interface of the model.

Much more land is cleared in a given year by existing ranches and farms than by new ones. Ongoing clearing rates are influenced by a clearing rate index. The clearing rate index is a function of the land speculation index, soil fertility (random–soil fertility is highly variable in the Amazon), erosion, and conflict. Conflict occurs between large and small landholders (and farmers and ranchers) in the absence of secure land tenure rights and title policies, and with increasing population density. Without working property rights institutions, an unofficial 'clear equals claim' policy drives farmers and ranchers to accelerate rates of deforestation.

Parameters were calibrated to generate clearing rates that are in line with those documented by INPE (1998). Roughly 30% of deforestation is estimated to come from agricultural (farm) clearing and 70% is attributed to pasture (ranch) clearing.

3.2. Land-use/cover sector

A land-use transition sector is shown in Fig. 3a and b. Fig. 3a shows part of the sector that deals with the transition rates and Fig. 3b shows calculation of biomass amounts in the land stocks. The transition rates were translated from Fearnside (1996), who used a first-order Markov model of transition probabilities between land-use categories to investigate carbon stocks in vegetation replacing the Amazon forest. Fearnside's approach was particularly helpful for our model because it explored the fate of land being cleared by both small farmers and ranchers, according to their typical behavior in terms of pattern of use, averaged time of use and of subsequent regrowth. Average and constant transition rates were weighted for small farmers and ranchers and then used as rates of transition probabilities between the land use categories. Although not necessarily realistic, Fearnside (1996) considered these ratios useful for estimating average biomass characteristics in each category. These averages are conservative: in reality, many of the economic pressures described in the 'drivers' sector of the model will force land to be used longer and more intensively, further reducing biomass. Fig. 4 shows the schematic diagram of land-transition derived from Fearnside (1996) and employed in this model.

We added to this transition model an annual flow of new deforested land derived from the deforestation drivers sector, which is a combination of new pasture clearing and new agriculture clearing. Over time, the model distributes the cleared land into the common categories found in the Brazilian Amazonia. The transition pattern begins with initial use as farmland (F) or productive pasture (PP), assumed in this model to correspond to smallholders and ranchers, respectively. Farmland transitions to either productive pasture or secondary forest (SFF). Productive pasture transitions into either degraded pasture (DP) or secondary forest from pasture (SFP). A small amount of secondary forest from farm and from pasture ends up in true succession to regenerated forest (RF). Most land, however, is continually transitioning between varying states of use and disuse (fallow), reflecting some of the true dynamics of land use in the Amazon. The transition values and state variables used in the model were estimated from Fearnside (1996), and approximate the land use dynamics that existed in the Amazon in 1990.

It is important to track different land-cover stocks in the model because different land use patterns greatly impact the quality and value of ecosystem services. One of the most important impacts of different land uses is their effect on biomass. High biomass productivity rates in the Brazilian Amazonia play a critical role in stabilizing and regulating ecological processes operating at local, regional and global scales. In the model, each land use stock was associated to average biomass properties according to a review of varying sources (Olson et al., 1983; Brown and Lugo, 1984; Uhl, 1987; Saldarriaga et al., 1988; Brown and Lugo, 1992; Fearnside, 1992a,b, 1996; Chroeder and Winjum, 1995; Salomao et al., 1996; Cochrane et al., 1999). A user interface allows model uses to adjust the biomass amounts in each stock to account for the range of values found in the literature. The initial forest biomass settings were derived as average numbers that take into account estimations for dense and non-dense forests (Fearnside, 1992a). In the model, the assumption is made that vegetation is uniform across the basin. Equations of land use transition (inflows and outflows rates), initial stock values, proportion of land in each category in relation to total deforested land, average biomass values and total biomass values and are provided in the Appendix A.

3.3. Ecosystems services sector

Ecosystem services refer to ecological conditions and processes that regulate and provide for human well being (Daily, 1997). This sector (shown in Fig. 5) focuses on four primary ecosys-

tem services that are provided for by an intact (i.e. forested) Amazonia region, and which contribute to human well-being on global, regional and local scales. They include climate regulation, erosion control, nutrient cycling and species diversity. Average ecological values for the assessed service and associated land use categories were derived from literature and are displayed in Table 1.

3.3.1. Climate regulation

The Amazon is the largest stand of tropical forest left on the planet and as such, it is an important carbon sink that aids in the maintenance of global climate regulation. In the model, we estimated carbon storage capacity to be 45% of the value of biomass (Fearnside, 1996). Storage capacity drops as biomass diminishes under farming and ranching land-use patterns. The Secondary succession has a smaller storage capacity than mature forest (ibid.), mainly due to the loss of large, mature trees (Attiwill, 1994).

In the model the carbon storage capacity of different land-use stocks is determined by their average rates of above-ground biomass as presented by Fearnside (1992a), Fearnside and Guimarães (1996) and Olson et al. (1983). The product of biomass, average carbon content, and square kilometers was calculated for each category. This result yielded carbon amounts in each land use stock. Total values of carbon were calculated by adding up the amount of carbon in all categories.

3.3.2. Erosion control

Hydrology and biomass are tightly connected in Brazilian Amazonia. Over 50% of precipitation in the region is due to water recycling through evapotranspiration (Salati, 1987; Zhang and Henderson-Sellers, 1996). Less biomass means less evapotranspiration and less precipitation. But with regard to the rain that does fall, less interception means that a higher percentage of total water volume falls directly onto the land surface, increasing surface runoff and erosion (Salati and Vose, 1984; Lavelle, 1987; Salati, 1987; Shukla et al., 1990; Fearnside, 1996). The hydrology process over a 100-year simulation is shown graphically in Fig. 6.

Average erosion rates in undisturbed forest were measured and reported by Salati and Vose (1984) to be approximately 116 tons km^2 $year^{-1}$. In the model a baseline erosion factor for forest is generated to be consistent with Salati's number by means of correlation to the biomass fraction, which is explicit in the hydrology submodel. Erosion rates and biomass are inversely related, and studies have found that erosion from the most degraded land is, on average, 7 times higher than erosion from forested land (Lavelle, 1987). Using this spectrum, and making the assumption that degraded pasture would have the highest erosion rate (7 times that of forest), average erosion rates are estimated for each type of land-use stock in the model. The model calculates total erosion figures associated with each type of land use. An erosion index that calculates the rising erosion rates as an index between 0 and 1 feeds back into the land clearing index in the deforestation drivers submodel.

3.3.3. Nutrient cycling

While nutrient levels in Amazonia ecosystems as a whole are high, nutrient cycling is relatively limited. The majority of nutrient stocks are accumulated in the standing biomass rather than in the soil (Salati and Vose, 1984; Lavelle, 1987). Nitrogen and phosphorous are exceptions to this rule and greater amounts are found in the soil, but the cation exchange capacity of soils is severely limited (Lavelle, 1987; Shlesinger, 1991). Clearing the standing biomass of rainforests for pasture and agriculture greatly reduces the nutrient cycling potential of the system (Reiners et al., 1994). Hecht (1983) asserts that with forest conversion to other uses, nutrients held in the biomass are shifted into soil nutrient storage, crops and weeds, or just lost through leaching and erosion. Hence, although a short period of time might follow where soils are actually enhanced by nutrients released in ash from the burning process, these nutrients are quickly leached out of the system (Hecht, 1983; Werner, 1984) due to increased runoff and erosion.

There is limited information available on nutrient amounts that exist above and below ground in the tropical forest (Vitousek, 1984). A review of

literature yielded a range of research results and a lack of consensus regarding the below-ground nitrogen storage capacity associated with varying types of land use. The range of research outcomes includes: (1) an initial increase in Nitrogen following clearing with a subsequent equilibration; (2) no significant differences before and after forest clearing; and (3) a decrease in soil mineral Nitrogen site after clearing (Eden et al., 1990; Neil et al., 1997; Hughes et al., 1999). Also, there is evidence that post-clearing treatment and land management practices are important factors in the soil chemical properties (Allen, 1985; Eden et al., 1990; Neil et al., 1997). As a general rule, however, after cutting and burning, soil levels of nitrogen are likely to drop (Ayanaba, 1976) as a result of volatilization (Hecht, 1983).

Brown and Lugo (1990) report decreases in soil Nitrogen pools as a result of forest conversion to pasture and cropland and accumulations of this nutrient as succession takes place. Furthermore, their study shows a pattern of increasing Nitrogen with increasing age of secondary forest and of decreasing Nitrogen with increasing soil depth. It also shows significant lower soil Nitrogen concentration under crops than under forest and pasture sites. This model uses Brown and Lugo's measurements of Nitrogen content values in the top 25 cm of soil as a proxy for nutrients. These Nitrogen values were selected because they account for land use intensity and ecosystem processes (i.e. carbon storage capacity, succession) that are closely aligned with the dynamics depicted in this model. An average Nitrogen content of a mature forest was estimated as 0.7 kg m^{-2}. Storage capacity was also measured for land that had been converted to pasture and farmland, as well as for that in succession following use in either category, and is listed in Table 1. Nitrogen values were multiplied by area to determine a total nutrient value for each land use stock.

3.3.4. Species diversity

Tropical forests cover only 6% of the earth's surface, but are home to over half of all species on the planet (Wilson, 1991). Brazilian Amazonia represents the largest contiguous area of tropical forest that is left on the planet, but this area is quickly diminishing due to deforestation and land cover change. Land transformation is the primary driver of biodiversity loss (Vitousek et al., 1997), and occurs in Amazonia primarily as a result of farm and ranch activity. Although the overall clearing patterns for pasture are many times larger than agricultural clearing activity, the clearing pattern for agriculture is more fragmented and contributes to severe edge effects which extend the area affected by agricultural clearing significantly (Laurance et al., 1998a,b). The edge effect can double the amount of area impacted by agricultural clearing, which has important implications for species loss (Lugo, 1988; Tilman, 1994).

In the model, species loss occurs as a result of changes in land cover (a proxy for changing habitat). Ranching and farming practices generate different types of clearing patterns and edge effects. The relationship between percentage of deforested land and related affected area by edge effect is graphed for both farm and pasture (Laurance et al., 1998b). Next, proportion P of species loss because of habitat destruction is defined as Eq. (1) (Tilman, 1994), where D is total area affected (deforested plus edge effect), and z is a constant.

$$P = 1 - (1 - D)^z \tag{1}$$

This equation was calibrated to mimic extinction of species in tropical forests as predicted by Lovejoy (1980), Ehrlich and Ehrlich (1981), and as cited by Lugo (1988). These authors estimated that species present in Latin America vary between 300 000 and 1 million. Their conservative projection of 50% deforestation corresponded to a 33% loss of species.

4. Ecosystem valuation submodel

This submodel associates the loss in ecosystem services related to conversion of forest to farm and pasture with a monetary reference value per hectare. 'Farm value' incorporates active farmland (F) and secondary forest from farm (SFF). "Pasture value" includes productive pasture (PP), degraded pasture (DP) and secondary forest from pasture (SFP).

The original monetary value of ecosystem services used as reference for tropical forest are taken from a 1997 publication, 'The value of the world's ecosystem services and natural capital' (Costanza et al., 1997). These values are shown in Table 2. The total per hectare value of these services for tropical areas is reported to be \$1431 annually. Farm values and pasture values calculated by the model represent the decreased ecosystem service value of land that has been cleared for these productive uses. Values are calculated for four primary services: (1) climate regulation; (2) nutrient cycling; (3) erosion control; and (4) genetic resources.

4.1. Climate regulation

Climate regulation is based on carbon storage capacity. In the valuation of carbon, the average carbon content of biomass in each land category (45% of above-ground biomass according to Fearnside and Guimarães, 1996), was weighted by the average carbon amount in forest and then multiplied by the forest monetary value for that service (\$223 ha^{-1} $year^{-1}$). Calculations were done to obtain total carbon value (Total C value), which is an annual monetary value for the entire Brazilian Amazonia resulting from the aggregated area of farm (Farm carbon value), pasture (Pasture carbon value) and forest (Forest carbon value). A unit value for each land category was also calculated (unit Farm C, and unit Pasture C). This value corresponds to an annual flow of service per hectare of land use, and is useful in comparing with the Costanza's reference value for climate regulation.

4.2. Nutrient cycling

Nutrient cycling is based on the amount of Nitrogen present in soil nutrient pools. These values were explained in detail in the ecosystem services section of this paper. The amount of Nitrogen present in primary forest samples was associated to the value for nutrient cycling (\$922 ha^{-1} $year^{-1}$) given by Costanza et al. (1997). The same process used to derive carbon values was used to calculate total and unit values for nitrogen present in different land use stocks.

4.3. Erosion control

The value for erosion control decreases as the amount of erosion in the ecosystem services submodel increases. The initial value for erosion control in primary forest was calculated by Costanza et al. at \$245 ha^{-1} $year^{-1}$. The erosion control values are weighted by the proportion of land that is incorporated in aggregated farm and aggregated pasture land use types.

4.4. Genetic resources

The ecosystem service value of genetic resources that is used in this model incorporates only the pharmaceutical value associated with species diversity (Costanza et al., 1997). Depreciation of this value in the model occurs as the overall number of species fall due to deforestation and increased edge effects. The original forest value for genetic resources is \$41 ha^{-1} $year^{-1}$.

5. Results and discussion

Results of a 100-year simulation run of the model show that forest area declines to about 44% of original forest area with pasture and abandoned pasture becoming the dominant land cover. The value of the four ecosystem services represented in this model declines for converted forest, from \$1431 to \$657 and \$781 ha^{-1} $year^{-1}$ for agriculture and pasture, respectively. Table 2 summarizes the findings, which are considered in more detail in the sections below.

5.1. Land use

In a 100-year modeling scenario, the Brazilian Amazonia forest area declines by 56%, and the total area ever deforested reaches 66%. This number fluctuates due to random parameters within the model, but consistently produces results within an acceptable range around this value. As is shown in Table 3, the majority of cleared land is either under use as productive pasture or is secondary forest derived from pasture. Taken together, these two categories account for 86% of

deforested land. Total pasture area, including the above two categories just noted and degraded pasture, account for 90% of land cleared. This is equivalent to 2 373 714 km^2. These results are consistent with results found in Fearnside (1996).

Farm and secondary forest from farm account for only 7% of total deforested land. This result may seem inconsistent with earlier observations that 30% of deforestation is at the hands of small farmers. Farmers, however, convert a large majority of their holdings into pasture as household structure, economic incentives and soil fertility change. This land becomes considered as pasture in future transitions in the model.

While we realize that there are tradeoffs and limitations to using static transition percentages to model dynamic land use patterns, we agree with Fearnside that such an approach is 'valuable as a first approximation' to dealing with the issue (ibid.). We were also inclined to pursue this option due to a lack of data with which to calibrate or track land use transitions using a more dynamic approach.

Land transitions and final amounts of land accumulating in different land use types are important for all of the changes in ecosystem services and corresponding decreases in monetary value. Results for each service are discussed in turn. Graphs of each service are located in Appendix B.

5.2. Carbon sequestration/climate regulation

Fifty-six percent deforestation of Amazonia over a 100-year model simulation resulted in a 42% decrease in carbon storage capacity. The discrepancy between deforestation and carbon storage projections is due to high level of secondary regrowth that occurs. This mitigates the loss in carbon storage capacity that would otherwise occur.

The value of climate regulation (related here as carbon storage capacity) per hectare decreases significantly between forested and deforested land. Values for carbon storage and climate regulation also differ between farm and pasture areas. Compared to the forest reference value of \$223 ha^{-1}, the value of climate regulation services for agricultural area in Amazonia falls to just \$7, and the value of pasture falls to \$11. The reason for the decline is the extreme loss of biomass associated with each land use type in the model. The average storage capacities were averaged between all farm-related land use stocks and all pasture-related land use stocks to derive these numbers. Farm area loses more carbon storage value per hectare than pasture due to the quicker transition periods that exist between use and fallow. Quick succession and higher land use intensity result in biomass levels that are lower over time.

The low average value given by Fearnside for biomass on agricultural land is the primary reason for the difference between farm and pasture. Given that leaf area indices for agricultural areas can be quite high, it is possible that this number is too low and loss of carbon storage value is overestimated.

5.3. Erosion/erosion control

The sediment load associated with deforested conditions is 1 billion tons $year^{-1}$. Without deforestation, this load is 572 million tons $year^{-1}$. This represents an erosion rate that is 2.4 times higher under deforestation simulation than would be the case without deforestation. The reference value for erosion control that we used was \$245 ha^{-1}. The corresponding values for farm and pasture land were \$66 and \$61, respectively. Pasture lost more value relative to farm primarily due to some of the model assumptions, including higher rates of erosion for degraded pasture than any other land use stock.

5.4. Nitrogen/nutrient cycling

Nitrogen storage capacity decreased in the model by 16% for the region as a whole during the 100-year simulation. The reference value for nutrient cycling services in tropical forests was quite high, \$922 ha^{-1}. Land used primarily as farm provided \$556 dollars of nutrient cycling services, a 40% reduction over the reference case. Pasture value was 27% lower than the reference forest value or \$677 ha^{-1}. The different values calculated for pasture and farm again appear to

be accounted for by differences in vegetation regrowth patterns between the two land use types. Farm areas are subject to greater nutrient leaching levels than are productive or secondary pasture due to the higher intensity of land use on farm property.

We see two limitations to our approach to modeling and valuing nutrient cycling services. First, the use of nitrogen storage capacity as a proxy for nutrient cycling is a great simplification of the nutrient cycling processes that occur in the Amazon. Second, the use of average numbers for nitrogen stocks in the land use categories creates limitation on the degree of feedback and dynamic behavior that can be reflected in the model.

5.5. Species diversity/genetic resources

Species loss in the Brazilian Amazonia grew to 51% over the 100-year simulation period, up from 34% in the initial scenario in the 1990 baseline year. This majority of species loss in the model was attributed to deforestation from pasture. This is not because—hectare per hectare—land converted to pasture is more damaging to species than land converted for farming. On the contrary, research indicates that the edged effects and intensity of land use associated with farming creates more threats to species diversity than large scale ranching. In an aggregated model such as this, however, the overwhelming scale of deforestation related to pasture use means that more species loss will be attributed to this type of land use than to farming.

The reference value for genetic resources is \$41 ha^{-1} (Costanza et al., 1997). On a hectare by hectare basis, farmland lost more of its genetic resource service value than ranch land. The model showed that the annual service value of genetic resources on farm land fell to \$29 ha^{-1} $year^{-1}$, while that of pasture fell to \$32 ha^{-1} $year^{-1}$. The reference value for genetic resources only refers to the market value for pharmaceuticals. Species diversity, however, has been shown by many ecologists to play a larger role in ecosystem stability (Holling, 1996; Peterson et al., 1998). For this reason, we believe that the value of genetic resources undervalues the total contribution of species diversity to ecosystem services.

5.6. Comparison of market and ecosystem service values

Addition of the adjusted values for climate regulation, nutrient cycling, erosion control and species diversity numbers reveal that the overall per hectare value of ecosystem services declines by 45% for ranching, and by 54% for farming over the period of simulation. The difference between the two rates of depreciation stems mainly from the high monetary reference value ascribed to nutrient cycling as a service, and the fact that agricultural practices tend to cause greater disruption of this cycle.

Investigating the extent to which different land use practices and patterns of land cover change degrade the monetary value of ecosystem services is a helpful process in its own right. The altered value of ecosystem services becomes even more of a discussion point, however, when it is compared to the annual revenue streams that flow from the land use practices that replace forest and cause the depreciation in their service value.

In recent years, initial efforts have been made to calculate the revenue generated by ranching and agricultural practices in the Brazilian Amazonia. A series of studies designed by Christopher Uhl and research partners were conducted to document the average annual income per hectare that widespread ranching and farming techniques generate (Mattos and Uhl, 1994; Toniolo and Uhl, 1995; Almeida and Uhl, 1995). Their research documents gross annual returns, profits, investment costs and other calculations that pertain to the prevailing extensive models of both ranching and agriculture. For purposes of this analysis, we have chosen to present the annual value of ecosystem services alongside the gross annual returns to ranching and farming presented by Almeida and Uhl (1995).

For prevailing models of extensive ranching, gross returns are calculated to be \$31 ha $year^{-1}$. For prevailing models of agricultural production (extensive and shifting), returns are calculated at \$90 ha $year^{-1}$. When gross returns from ranching and farming are compared to the annual value of ecosystem services, the disparity is striking: a gross annual return to ranching of \$31 ha^{-1}

compares to an ecosystem service value of \$781 ha^{-1}. Similarly, a gross annual return to agriculture of \$90 ha^{-1} compares to an ecosystem service value of \$658 ha^{-1}. Even with the significant monetary losses in the value of ecosystem services over the forest reference value of \$1431, the annual value of services from land used for ranching is 25 times the amount of revenue generated from a hectare of land used for ranching. The value of ecosystem services provided by land used for farming is 7 times greater than the revenue farmers can generate from their activities. If land was kept entirely out of production and remained as undisturbed forest, the differential between the annual service value of the ecosystem and the annual revenue from ranch and farm activities would be 48 and 16, respectively.

It is quite possible that the depreciation in ecosystem services values generated by the model is too conservative. The assumption is that the value of ecosystem services decreases in a linear fashion. In biological systems, however, there is a point where the degree of degradation compromises the system's stability and its overall resilience (Holling, 1996; Peterson et al., 1998). In such cases, the services (and thus the corresponding monetary value of the service) may be irretrievably lost (Barbier, 1994).

There are obvious practical problems associated with comparing the annual monetary flows of ranching and farming activities with the inferred value of ecosystem services. Such comparisons are irrelevant to individuals engaged in ranching and farming because ecosystem services are public goods (Lawn, 2001) and, as such, carry no 'real' monetary value that individuals could benefit from. While there is a private monetary return to individuals who engage in ranching and farming practices, the existence of a pristine forest provides services that benefit society as a whole. The current non-market—and hence non-priced—nature of ecosystem services is an impediment to creating a system of incentives that would lead land holders in Brazilian Amazonia to see a loss in the value of ecosystem services as significant opportunity cost. Under the conditions simulated in this model, the opportunity costs associated with converting forest into ranching and farming uses would be \$650 and \$773 ha^{-1} $year^{-1}$, respectively.

6. Conclusion: toward a rationale for explicit valuation of ecosystem services

The model described in this paper provides a rough approximation of the loss of ecosystem services that is attributable to deforestation, if current patterns and processes of land use—and the economic incentives that drive them—continue unabated.

Land usage in the Brazilian Amazonia is currently influenced by an array of private preferences and public mandates. On one side, individual ranchers and farmers work to ensure their well-being by using their land in ways that generate the highest returns. At the same time, the public needs a healthy ecosystem to regulate regional and global climate patterns, and provide other fundamental services that ensure well-being. As we have demonstrated with this model, the private preferences of individuals are not always compatible with public needs (Norgaard, 1989).

The monetary approach to ecosystem valuation provides one means of overcoming the incompatibility of public and private preferences. When the forests that provide vital services are valued by private markets in monetary terms, individuals receive signals that indicate the importance of resource conservation. What a monetary valuation of ecosystem services cannot convey, however, is a sense of the intrinsic or inherent value of an intact ecosystem that exists regardless of human benefit.

This model has been a first attempt to dynamically describe and display the processes that degrade ecosystem services over time. In the future, more work must be done to provide alternative scenarios that can demonstrate ways in which development of Amazonia can be sustaining for both the people who live in Amazonia and people from other parts of the world who depends on the services provided by this ecosystem. Assessment of the value of ecosystem services is crucial in bringing awareness and understanding to the benefits they provide, and to the absolute need of

taking such values into account in decision-making processes.

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Appendix A

A.1. Deforestation drivers sector

```
ag_households(t) = ag_households(t − dt)
                   + (farms_per_year)*dt

INIT ag_households = 1 445 142

  Inflows:

farms_per_year = new_ag_households

Population(t)
= Population(t − dt)
  + (Pop_growth-outmigration_and_death)*dt

INIT Population = 9 337 153

  Inflows:

Pop_growth
    = ((Population*0.02) + migrants)
      *(1 − Population/3e7)

  Outflows:

outmigration_and_death = 0.005*Population

ranches(t)
= ranches(t − dt) + (ranches_per_year)*dt

INIT ranches = 156 399

  Inflows:

ranches_per_year = new_ranches

ag_population = 0.45*Population

ag_pop_change = ag_population
                − DELAY(ag_population,1)

Amplitude = 25

Base_year = 20

Economic_growth_rate = 2

Economic_trend
  = DELAY((Econ_Long_term_trend
  + Econ_periodicity + Econ_random), 0.5)

Econ_Long_term_trend
    = 25 + ((TIME − 1990)*
      Economic_growth_rate)

Econ_periodicity
    = (SINWAVE(Amplitude, Period))

Econ_random = RANDOM( − 15,15)

Econ_trend_index
    = (Economic_trends/Base_year)/100

land_spec_index
    = (Econ_trend_index) + Infrastructure

migrants = ag_pop_change*migration_rate

migration_rate = (land_spec_index
                 *Non_Amazon_Land_Dist)*
                 (new_ranches*0.05)

new_ag_households
    = (ag_pop_change + migrants)/5
```

new_ranches = land_spec_index*property
–investors

Non_Amazon_Land_Dist = 0.05

Period = 10

property_investors
= (2*ranches/Biomass_fraction)*100

Growth_Data = GRAPH(time)

(1984, 2.00), (1985, 1.90), (1986, 2.00),
(1987, 2.00), (1988, 1.90), (1989, 1.80),
(1990, – 4.00), (1991, 1.00), (1992, – 1.00),
(1993, 5.00), (1994, 5.80), (1995, 4.30),
(1996, 2.90), (1997, 3.50), (1998, 0.2)

Infrastructure = GRAPH(time)

(1990, 0.01), (1998, 0.09), (2007, 0.075),
(2015, 0.1), (2023, 0.11), (2032, 0.11),
(2040, 0.12), (2048, 0.205), (2057, 0.225),
(2065, 0.24), (2073, 0.265), (2082, 0.28),
(2090, 0.3)

A.2. Ecosystem services

Water(t) = Water(t – dt)
+ (Precipitation – Runoff – ET)*dt

INIT Water = 1.2*12.7e12

Inflows:

Precipitation = Rate_of_precipitation*
Water_vapor

Outflows:

Runoff = 0.8*Water*Erosion_factor

ET = Water*Evapotranspiration

Water_vapor(t) = Water_vapor(t – dt)
+ (Ocean_vapor + ET – Precipitation)*dt

INIT Water_vapor = Ocean_vapor + ET

Inflows:

Ocean_vapor = 5E12

ET = Water*Evapotranspiration

Outflows:

Precipitation = Rate_of_precipitation*
Water_vapor

ADPN = 400

AFN = 350

AFORESTN = 700

All_Farm = SFF + F

All_Forest = Pre1970_SF + RF + Total_forest

All_Pasture = DP + PP + SFP

Amazon_surface = 4E6

APPN = 450

APRE1970N = 700

ARFN = 0.7e3

ASFFN = 600

ASFPN = 600

Average_RF_biomass_2 = 148E2

Biomass_fraction = Total_biomass/
Amazon_surface

Carbon_index = ((Total_carbon –
Farm_carbon – Pasture_carbon)/Total_carbon)

Erosion_DP = 7*(Erosion_factor*DP)*100

Erosion_F = 4*(Erosion_factor*F)*100

Erosion_factor
= (1/(Biomass_fraction − 2500) ∧ 0.2
+ 1)

erosion_index = 1 − ((Total_erosion − Farm_erosion − Pasture_erosion)/Total_erosion)

Erosion_PP = 5*(Erosion_factor*PP)*100

Erosion_pre1970
= (Erosion_factor*Pre1970_SF)*100

Erosion_RF = (Erosion_factor*RF)*100

Erosion_SFF = 3*(Erosion_factor*SFF)*100

Erosion_SFP = 2.5*(Erosion_factor*SFP)*100

Erosion__Total_forest
= (Erosion_factor*Total_forest)*100

Evapotranspiration = Biomass_fraction/
(Biomass_fraction + 28 369)

Farm_carbon = (F_biomass + SFF_biomass)
*0.45

Farm_erosion = Erosion_F + Erosion_SFF

Farm_Nitrorgen = Nitrogen_F + Nitrogen_SFF

Forested_carbon = (Forest_biomass
+ Pre1970_biomass + RF_biomass)*0.45

Forest_erosion = (Erosion_pre1970
+ Erosion_RF + Erosion_Total_forest)

Forest_Nitrogen = Nitrogen_RF +
Nitrogen_Forest + Nitrogen_Pre1970

Nitrogen_DP = DP*ADPN

Nitrogen_F = F*AFN

Nitrogen_Forest = Total_forest*AFORESTN

Nitrogen_index = ((Total_Nitrogen
− Pasture_Nitrogen − Farm_Nitrorgen)/
Total_Nitrogen)

Nitrogen_PP = PP*APPN

Nitrogen_Pre1970 = Pre1970_SF*APRE1970N

Nitrogen_RF = RF*ARFN

Nitrogen_SFF = SFF*ASFFN

Nitrogen_SFP = ASFPN*SFP

Pasture_carbon = (DP_biomass
+ PP_biomass + SFP_biomass)*0.45

Pasture_erosion = (Erosion_DP + Erosion_PP
+ Erosion_SFP)

Pasture_Nitrogen = Nitrogen_DP
+ Nitrogen_PP + Nitrogen_SFP

Percentage_of_Farm_deforestation
= (F + SFF)/Amazon_surface

Percentage_of_Pasture_deforestation
= (DP + PP + SFP)/Amazon_surface

Rate_of_precipitation = RANDOM(0.5, 0.9)

Species_loss_by_farm
= 1 − (1 − Farm_edge_effecton_forest)
∧ 1.5

Species_loss_by_pasture
= 1 − (1 − Pasture_edge_effect) ∧ 1.5

Total_carbon = Farm_carbon
+ Forested_carbon + Pasture_carbon

Total_erosion = Farm_erosion
+ Forest_erosion + Pasture_erosion

Total_Nitrogen = Farm_Nitrorgen
+ Forest_Nitrogen + Pasture_Nitrogen

Total_species_loss = Species_loss_by
_farm + Species_loss_by_pasture

Farm_edge_effecton_forest = GRAPH (Percentage_of_Farm_deforestation)
(0.00, 0.165), (0.111, 0.27), (0.222, 0.32), (0.333, 0.36), (0.444, 0.365), (0.556, 0.365), (0.667, 0.36), (0.778, 0.345), (0.889, 0.305), (1.00, 0.17)

Pasture_edge_effect = GRAPH (Percentage_of_Pasture_deforestation)
(0.00, 0.025), (0.1, 0.075), (0.2, 0.11), (0.3, 0.14), (0.4, 0.16), (0.5, 0.16), (0.6, 0.15), (0.7, 0.13), (0.8, 0.08), (0.9, 0.04)

A.3. Ecosystem valuation

Average_Farm_Erosion = Farm_erosion/All_Farm

Average_Pasture_Erosion = Pasture_erosion/All_Pasture

Farm_C_value = All_Farm*Unit_Farm_C

Farm_E_value = All_Farm*Unit_Farm_E

Farm_N_Value = (Unit_Farm_N*All_Farm)

Farm_S_value = All_Farm*Unit_Farm_S

Forest_C_value = All_Forest*Unit_Forest_C

Forest_E_value = All_Forest*Forest_Value

Forest_N_value = All_Forest*Unit_Forest_N

Forest_S_value = All_Forest*Unit_Forest_S

Forest_Value = 245

Pasture_C_value = All_Pasture*Unit_Pasture_C

Pasture_E_value = All_Pasture*Unit_Pasture_E

Pasture_N_value = All_Pasture*Unit_Pasture_N

Pasture_S_value = All_Pasture*Unit_Pasture_S

Total_Amazon_Value = (Total_C_value + Total_N_Value + Total_E_value + Total_S_Value)

Total_C_value = (Farm_C_value + Forest_C_value + Pasture_C_value)*1e2/1e6

Total_E_value = (Farm_E_value + Forest_E_value + Pasture_E_value)*1e2/1e6

Total_N_Value = (Farm_N_Value + Pasture_N_value + Forest_N_value)*1e2/1e6

Total_S_Value = (Pasture_S_value + Farm_S_value + Forest_S_value)*1e2/1e6

Total_Unit_Farm = Unit_Farm_C + Unit −Farm_N + Unit_Farm_S + Unit_Farm_E

Total_Unit_Pasture = Unit_Pasture_C + Unit_Pasture_E + Unit_Pasture_N + Unit_Pasture_S

Unit_Avg_Value_S = (Farm_S_value/All_Farm + Pasture_S_value/All_Pasture)/2

Unit_DP_Carbon = ADPB*223/(AFORESTB)

Unit_DP_Nitrogen = ADPN*922/AFORESTN

Unit_Farm_C = ((F_biomass + SFF_biomass)/All_Farm)*223/(AFORESTB)

Unit_Farm_E = (Forest_Value*116)/Average_Farm_Erosion

Unit_Farm_N = (Nitrogen_F + Nitrogen_SFF)/All_Farm*922/AFORESTN

Unit_Farm_S =
(41 − (Species_loss_by_farm*41))

Unit_Forest_C = (Forest_biomass
+ Pre1970_biomass + RF_biomass)/
All_Forest*223/(AFORESTB)

Unit_Forest_N = (Nitrogen_Forest +
Nitrogen_Pre1970 + Nitrogen_RF)/
All_Forest*922/AFORESTN

Unit_Forest_S = 41

Unit_F_Carbon = AFB*223/(AFORESTB)

Unit_F_Nitrogen = AFN*922/AFORESTN

Unit_Pasture_C = ((DP_biomass
+ PP_biomass + SFP_biomass)/
All_Pasture)*223/(AFORESTB)

Unit_Pasture_E = Forest_Value*116/
Average_Pasture_Erosion

Unit_Pasture_N = (Nitrogen_DP
+ Nitrogen_PP + Nitrogen_SFP)/
All_Pasture*922/AFORESTN

Unit_Pasture_S =
(41 − (Species_loss_by_pasture*41))

Unit_PP_Carbon = APPB*223/(AFORESTB)

Unit_PP_Nitrogen = APPN*922/AFORESTN

Unit_SFF_Carbon
= ASFFB*223/(AFORESTB)

Unit_SFF_Nitrogen
= ASFFN*922/AFORESTN

Unit_SFP_Carbon = ASFPB*223/(AFORESTB)

Unit_SFP_Nitrogen
= ASFPN*922/AFORESTN

Unit_Value_S_loss
= (Unit_Pasture_S + Unit_Farm_S)

A.4. Land use/cover sector

DP(t) = DP(t − dt) + (Conversion_of_PP_
to_DP − Conversion_of_DP_to_SFP −
Conversion_of_DP_to_PP)*dt

INIT DP = 8E3

Inflows:

Conversion_of_PP_to_DP
= PP*Rate_of_PP_to_DP

Outflows:

Conversion_of_DP_to_SFP
= DP*Rate_of_DP_to_SFP

Conversion_of_DP_to_PP
= DP*Rate_of_DP_to_PP

F(t) = F(t − dt) + (Conversion_of_RF_to_F
+ Conversion_from_SFP_&_SFF
+ Conversion_of_deforested_land_to_Farming
− Conversion_of_F_to_PP
− Conversion_of_F_to_SFF)*dt

INIT F = 22E3

Inflows:

Conversion_of_RF_to_F
= RF*Rate_of_INIT F = 22E3

Conversion_from_SFP_&_SFF =
Conversion_of_SFP_to_F
+ Conversion_of_SFF_to_F

Conversion_of_deforested_land_to_Farming
= Rate_of_deforested_land_to_F*
New_deforested_land
Outflows:
Conversion_of_F_to_PP = F*
Rate_of_F_to_PP
Conversion_of_F_to_SFF = F*
Rate_of_F_to_SFF
PP(t) = PP(t − dt)
+ (Conversion_from_F_DP_SFP_SFF
+ Conversion_of_RF_to_PP
+ Conversion_of_deforested_land_to_PP
− Conversion_of_PP_to_DP
− Conversion_of_PP_to_SFP)*dt
INIT PP = 184E3
Inflows:
Conversion_from_F_DP_SFP_SFF
= Conversion_of_F_to_PP
+ Conversion_of_SFF_to_PP
+ Conversion_of_SFP_to_PP
+ Conversion_of_DP_to_PP
Conversion_of_RF_to_PP = RF*
Rate_of_RF_to_PP
Conversion_of_deforested_land_to_PP
= Rate_of_deforested_land_to_PP*
New_deforested_land
Outflows:
Conversion_of_PP_to_DP
= PP*Rate_of_PP_to_DP
Conversion_of_PP_to_SFP
= PP*Rate_of_PP_to_SFP
Pre1970_SF(t) = Pre1970_SF(t − dt)
INIT Pre1970_SF = 71e3
RF(t) = RF(t − dt) + (RF_conversion
− Conversion_of_RF_to_F
− Conversion_of_RF_to_PP)*dt
INIT RF = 0
Inflows:
RF_conversion = Conversion_of_SFF_to_RF
+ Conversion_of_SFP_to_RF
Outflows:
Conversion_of_RF_to_F
= RF*Rate_of_RF_to_F
Conversion_of_RF_to_PP
= RF*Rate_of_RF_to_PP
SFF(t) = SFF(t − dt)
+ (Conversion_of_F_to_SFF
− Conversion_of_SFF_to_F
− Conversion_of_SFF_to_PP
− Conversion_of_SFF_to_RF)*dt
INIT SFF = 8E3
Inflows:
Conversion_of_F_to_SFF
= F*Rate_of_F_to_SFF
Outflows:
Conversion_of_SFF_to_F
= SFF*Rate_of_SFF_to_F
Conversion_of_SFF_to_PP
= SFF*Rate_of_SFF_to_PP
Conversion_of_SFF_to_RF
= SFF*Rate_of_SFF_to_RF

SFP(t) = SFP(t − dt)
+ (Conversion_of_DP_to_SFP
+ Conversion_of_PP_to_SFP1
− Conversion_of_SFP_to_PP
− Conversion_of_SFP_to_RF
− Conversion_of_SFP_to_F)*dt

INIT SFP = 115E3

Inflows:

Conversion_of_DP_to_SFP
= DP*Rate_of_DP_to_SFP

Conversion_of_PP_to_SFP1
= Conversion_of_PP_to_SFP

Outflows:

Conversion_of_SFP_to_PP
= SFP*Rate_of_SFP_to_PP

Conversion_of_SFP_to_RF
= SFP*Rate_of_SFP_to_RF

Conversion_of_SFP_to_F
= SFP*Rate_of_SFP_to_F

Rate_of_forest_to_deforested
= New_deforested_land

OUTFLOW FROM:

Total_forest(Not in a sector)

INFLOW TO: Deforested_SV(Not in a sector)

ADPB = 3.4E2

AFB = 1E2

AFORESTB = 272

APPB = 10E2

APRE1970B = 148E2

ARFB = 148E2

ASFFB = 29E2

ASFPB = 17e2

DP_biomass = DP*ADPB

established_ag_clearing
= ((ag_households/12)
*(yearly_avg_clear_ag))
+ ((ag_households/12)*(clearing_rate_index))

established_pasture_clearing
= ((ranches/5)
*(yearly_avg_clear_ranch))
+ ((ranches/5*clearing_rate_index))

Forest_biomass = Total_forest*AFORESTB

F_biomass = AFB*F

new_ag_clearing
= ((new_ag_households*3E − 3)
+ established_ag_clearing)

New_deforested_land = (new_ag_clearing
+ new_pasture_clearing)

new_pasture_clearing
= DELAY(((new_ranches*50E − 2)
+ established_pasture_clearing),0.5)

Percent_of_Amazon_Def_Land
= Total_deforested_land/Original_Forest

PP_biomass = PP*APPB

Pre1970_biomass = Pre1970_SF*APRE1970B

Proportion_of_DP = DP/Total_deforested_land

Proportion_of_F = F/Total_deforested_land

Proportion_of_PP = PP/Total_deforested_land

Proportion_of_pre1970_SF = Pre1970_SF/
Total_deforested_land

Proportion_of_RF = RF/Total_deforested_land

Proportion_of_SFF = SFF/
Total_deforested_land

Proportion_of_SFP = SFP/
Total_deforested_land

Rate_of_deforested_land_to_F
= new_ag_clearing/New_deforested_land

Rate_of_deforested_land_to_PP
= new_pasture_clearing/New_deforested_land

Rate_of_DP_to_PP = 0.007

Rate_of_DP_to_SFP = 0.067

Rate_of_F_to_PP = 0.468

Rate_of_F_to_SFF = 0.082

Rate_of_PP_to_DP = 0.008

Rate_of_PP_to_SFP = 0.143

Rate_of_RF_to_F = 0.347

Rate_of_RF_to_PP = 0.653

Rate_of_SFF_to_F = 0.065

Rate_of_SFF_to_PP = 0.128

Rate_of_SFF_to_RF = 0.000000001

Rate_of_SFP_to_F = 0.061

Rate_of_SFP_to_PP = 0.101

Rate_of_SFP_to_RF = 0.0000001

RF_biomass = RF*ARFB

SFF_biomass = SFF*ASFFB

SFP_biomass = SFP*ASFPB

soil_fertility = RANDOM(0.1, 1)

Total_biomass = Forest_biomass
+ Total_deforested_area_biomass

Total_deforested_area_biomass
= RF_biomass + F_biomass
+ PP_biomass + SFF_biomass
+ DP_biomass + SFP_biomass
+ Pre1970_biomass

Total_deforested_land = (DP + F + PP
+ Pre1970_SF + RF + SFF + SFP)

yearly_avg_clear_ag = 0.3e − 2

yearly_avg_clear_ranch = 25e − 2

Deforestation_INPE = GRAPH(time)

(1988, 22 530), (1989, 23 900), (1990, 13 800),
(1991, 11 200), (1992, 13 790), (1993, 14 900),
(1994, 14 900), (1995, 27 080), (1996, 20 010),
(1997, 13 230), (1998, 16 840)

A.5. Not in a sector

Deforested_SV(t) = Deforested_
SV(t − dt) + (Rate_of_forest_to_deforested)*dt

INIT Deforested_SV = 410e3

Inflows:

Rate_of_forest_to_deforested(IN SECTOR:
Land use/cover sector)

Total_forest(t) = Total_forest(t − dt)
+ (− Rate_of_forest_to_deforested)*dt
INIT Total_forest = 4E6 − 410e3

Outflows:

Rate_of_forest_to_deforested(IN SECTOR: Land use/cover sector)

clearing_rate_index = (Conflict + soil_fertility + erosion_index + land_spec_index)/100

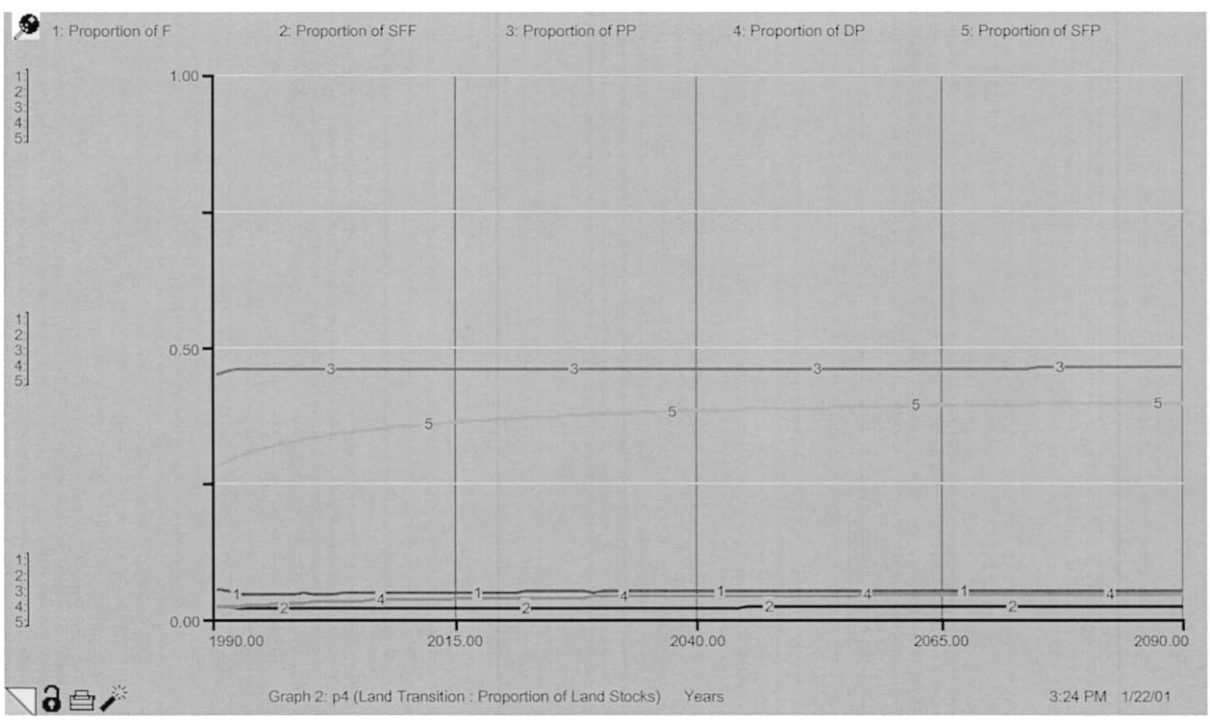
Conflict = If Tenure_security = 1 THEN 0 ELSE 0.03

Original_Forest = 4E6

percent_deforested = Deforested_SV/Total_forest

Tenure_security = 1

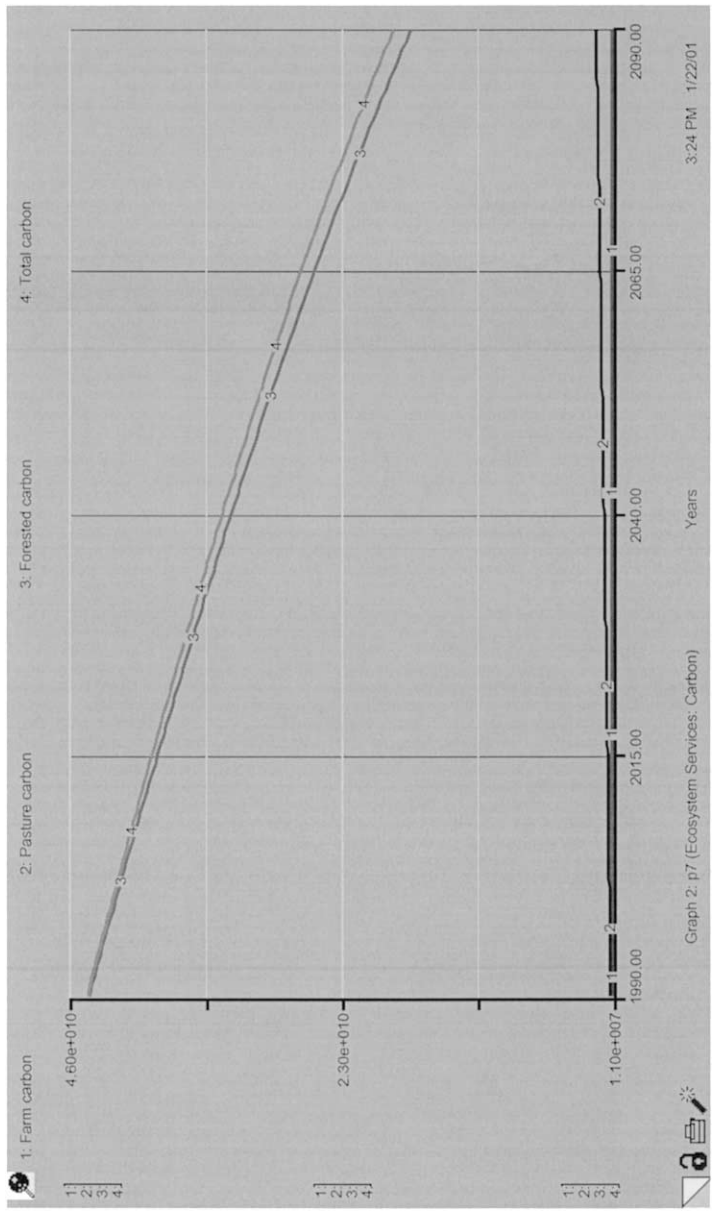
Appendix B



Proportion of land use under categories of Farm (F), Secondary Forest from Farm (SFF), Productive Pasture (PP), Degraded Pasture (DP), and Secondary Forest from Pasture (SFP).

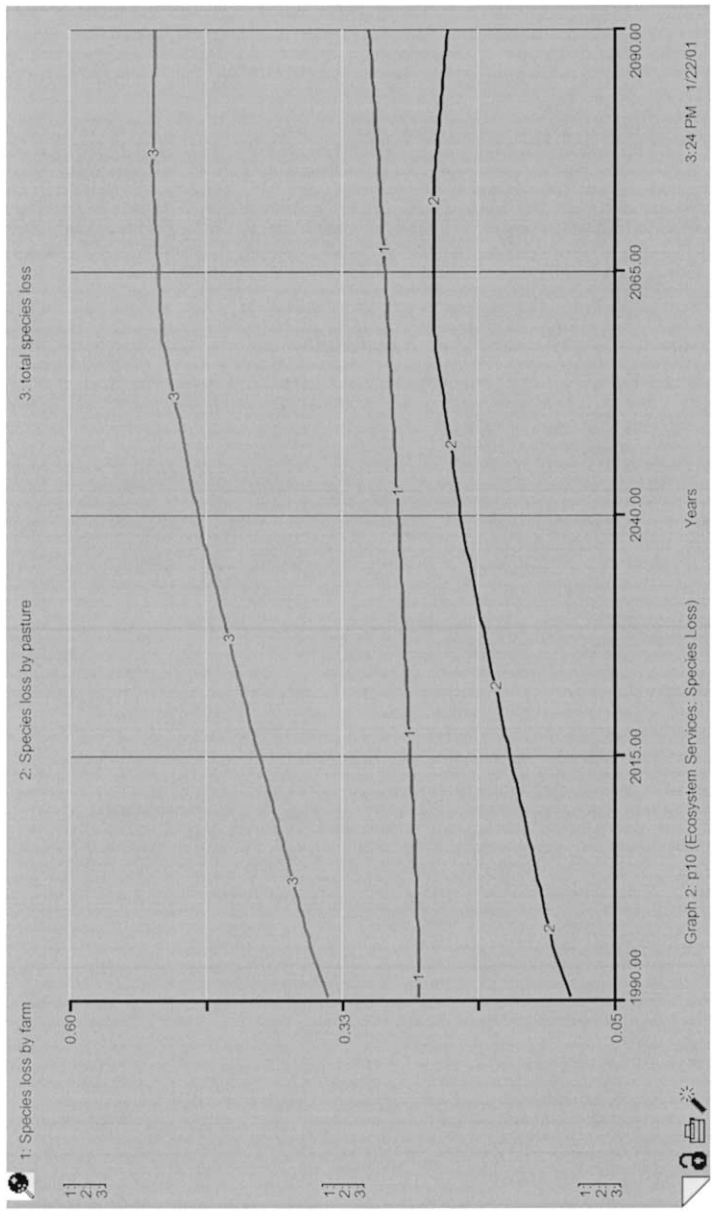
Appendix C

Carbon amounts (MT) on different land categories.



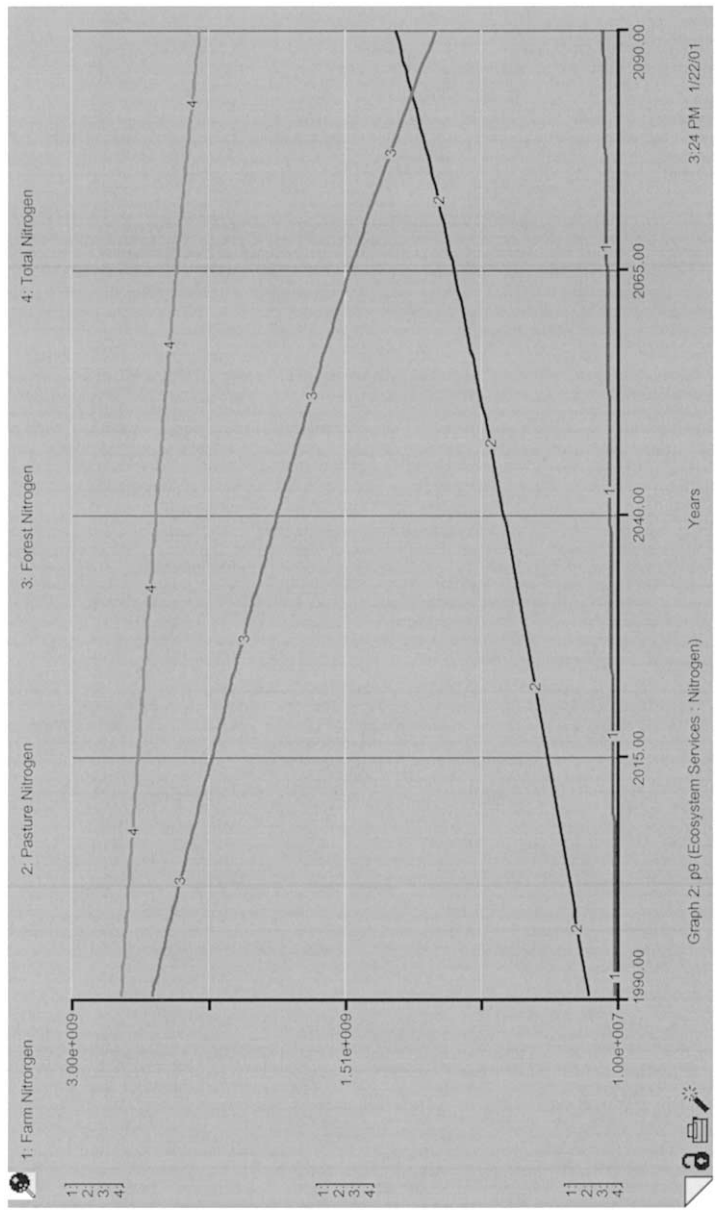
Appendix D

Erosion amounts (MT) on different land categories.



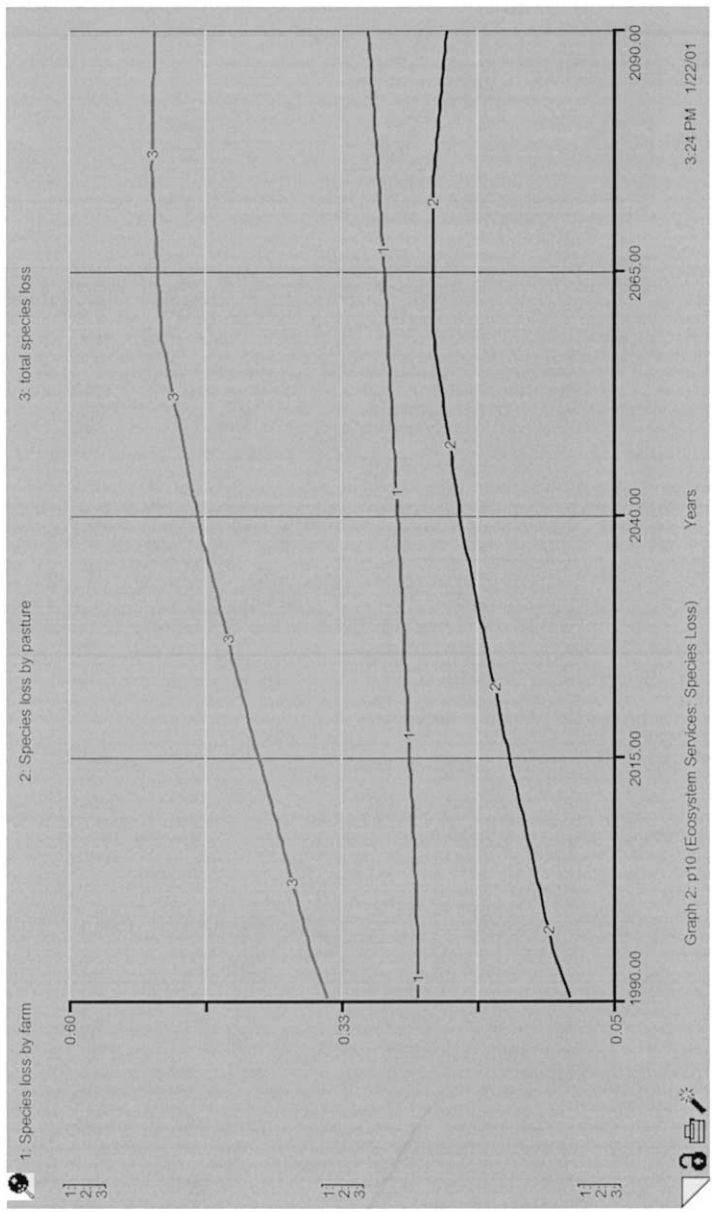
Appendix E

Nitrogen amounts (MT) on different land categories.



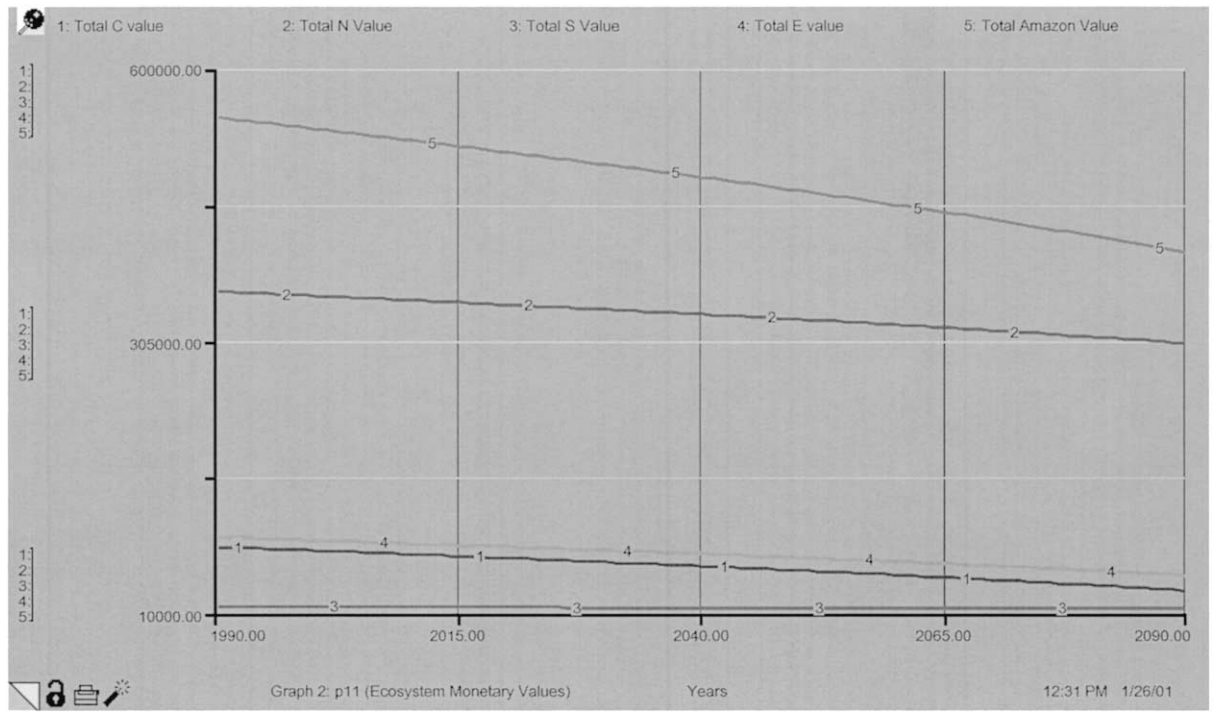
Appendix F

Species loss on different land categories.



Appendix G

Total monetary value (US$x1E6) of different services in the Brazilian Amazon.



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