ELSEVIER

Ecological Economics 18 (1996) 41–54

ECOLOGICAL
ECONOMICS

Analysis

Restoration of vegetation cover in the eastern Amazon

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Received 20 December 1995; accepted 19 February 1996

Abstract

Development activities in Altamira, Pará, Brazil began with the construction of the Transamazon Highway in 1970. In the early years land use emphasized annual crops, followed by pasture development. During the second decade of settlement, land cover in the region shifted from 'degraded pastures' or early stages of secondary successional vegetation to increased dominance of intermediate and late secondary succession. This finding is a result of recent research using Landsat TM satellite data and detailed field studies of vegetation stand structure. Close to 40% of the forest had been removed by 1985, but between 1985 and 1991 there is clear evidence for twice as much regrowth as there is new deforestation. Land use has a significant effect on the vegetation's total height, stem height, and basal area. The use of satellite data in combination with detailed studies of vegetation stand structure and land use histories gathered from households provide a multilevel approach that allows integration of local, regional, and global patterns of secondary succession in agricultural areas, and informs restoration strategies for Amazonia.

Keywords: Deforestation; Secondary growth; Vegetation; Restoration ecology; Policy implications; Satellite data; Soils; Fallows; Amazonia

1. Introduction

The devastating processes of deforestation in the Amazon Basin have attracted a great deal of attention in the past decade. The exponential rates of deforestation in the Brazilian Amazon in the 1975–1985 decade resulted in persistent international pressure and led to major policy shifts (Mahar, 1988; Anderson, 1990). One of these important shifts was the reduction in fiscal incentives that had encouraged the destruction of forests by supporting policies that did not value land until the forest cover was removed. The reduction in fiscal incentives for cattle ranching, together with the economic stagnation and hyperinflation that Brazil experienced from 1987 to 1994, has been associated with reduction in the rates of deforestation. With the reduction in hyperinflation and the return of positive rates of economic growth in 1995, deforestation rates have begun to increase once again, making it all the more urgent to understand the processes of deforestation in Amazonia.

Attention to these questions has deflected discussion of a no less important process in the region—the natural regrowth of vegetation following deforestation and the use of these secondary forests by farmers. This natural restoration of the landscape has important implications for processes such as the global carbon cycle, the hydrological cycle, and the sustainability of agricultural systems in the humid

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PII S0921-8009(96)00034-1

tropics (Uhl et al., 1990; Nepstad et al., 1991; Moran et al., 1994). Until recently it was impossible to monitor the difference between deforestation in primary and secondary forests in Amazonia. The inability to distinguish between primary and secondary forests bears considerable significance for land-use policy and for processes such as global warming. This natural regrowth is taking place not only in the Amazon, but elsewhere in the humid tropics. Lugo and Brown (1993) estimate that at least 250 million hectares are undergoing succession worldwide. Among the challenges to the study of tropical ecosystems are (1) to understand how vegetation changes over time as natural and human-induced disturbance occurs and (2) how to sample such areas so that an understanding of the dynamic spatial and temporal processes can inform economic and environmental policies. Little attention has been given to the processes of succession that are transforming large portions of deforested areas not into deserts, but into productive agricultural land and/or secondary successional vegetation cover. This process of succession can be quite rapid, with some areas being indistinguishable in terms of biomass and canopy structure from adjacent mature forest within 20 years. Such regrowth is rapid enough to make it difficult to locate selectively logged areas within 3 years (Stone, 1994) and abandoned pastures within 20 years (Mausel et al., 1993). This statement is not equivalent to saying that there are not significant differences in species diversity and stand composition between secondary forests and mature forests. What it does imply is that if sustainable development is to mean anything, it must be concerned with the management of successional processes by human populations. Following a period of intense initial deforestation, farmers in the Amazon frontier cut secondary growth more often than mature forests, especially when near roads which allow reasonable access to markets. Understanding the demographic, social, and economic conditions under which farmers turn to secondary, rather than primary, forests can mean the difference between sustainability and devastation.

This paper reports on studies carried out over the past 3 years in the eastern Amazon by a team of U.S. and Brazilian scientists, focusing on the processes of secondary succession that are gradually restoring forest cover to previously deforested areas. Between 1991 and 1993, the study focused on two relatively nutrient-rich sites.[1] This paper relies not only on traditional household surveys and studies of vegetation stand structure, but gives a central role to research tools such as satellite remote sensing and Geographic Information Systems (GIS) in assessing the changing landscape west of Altamira in the state of Pará, Brazil (see Fig. 1). Tools such as remote sensing, especially those satellite platforms or aerial photos that permit fine-grained analysis, provide a useful tool both for focusing at local scales, such as a forest gap or a small deforested area (Mausel et al., 1993), as well as permitting analysis of regions of many thousands of square kilometers—in this case of 2670 km^2 (Moran et al., 1994). The Altamira

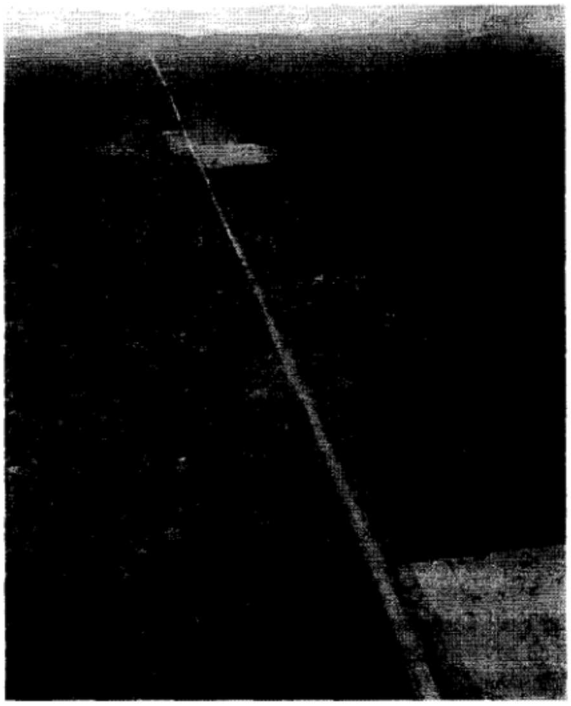

Fig. 1. The Altamira region was cut by the Transamazon highway. Remote sensing techniques make analysis of land cover change more comprehensive than traditional methods (Moran et al., 1994).

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[1] Our study has now expanded to three nutrient-poor sites: Igarapé-Açu, Pará; Tomé-Açu, Pará; and Yapú, in the Colombian Vaupés region.

region represents an 'aging frontier' that can provide insights into processes such as land-use intensification following a period of exponential deforestation.

2. Traditional approaches

Attention has centered on deforestation of primary forests in Amazonia not only because of the worldwide concern with the destruction of biodiversity, but also because of the technical difficulties of monitoring changes in vegetation in a region as vast as the Amazon. For analysis of very large regions such as the Brazilian Amazon, which requires more than 200 Landsat satellite scenes to cover the region (each scene being 185 by 185 km), it has been common to rely on single band Landsat photographic products [2] (Skole and Tucker, 1993) or, more frequently, on NOAA's AVHRR (1.1 km resolution) metereological satellites (Tucker et al., 1984; Woodwell et al., 1986; Matson and Holben, 1987).

Use of these data has permitted a number of scientists over the past 10 years to assess the extent of deforestation. Although NOAA's AVHRR was designed for metereological studies, it has been used to monitor vegetation patterns over very large areas. Specialists recognize that the spectral bands are not ideally positioned for vegetation analysis (Tucker et al., 1984). It cannot monitor changes in types of vegetation cover, but only distinguish between dense forest and its absence. It is useful for monitoring how many fires in a given day have been set in the Amazon Basin (the most dramatic data being for 1987 when on a given day about 8000 fires were recorded (Matson and Holben, 1987; Booth, 1989; Setzer and Pereira, 1991)). Several efforts to use AVHRR and the more refined scale of Landsat MSS (at 80 m spatial resolution) to observe secondary growth following deforestation met with no success (Woodwell et al., 1986, 1987). AVHRR data cannot be used to study specific sites, but only to provide overviews of general ecological conditions and to identify regions where more detailed study with higher resolution data may be required (Sader et al., 1990). Landsat MSS can be used to detect forest clearings at a scale of 1:200000, but it has only marginal utility for monitoring forest changes related to partial removal of the canopy (e.g., selective logging), successional changes, and other forms of canopy alterations (Sader et al., 1990, p. 1345). This same author found a lack of studies using Landsat Thematic Mapper satellite data, with 30 m resolution, and the French SPOT satellite, with 10–20 m resolution, in studying changes in the tropics. Landsat TM and SPOT provide information of considerable ecological richness for local and regional analysis, but are less suitable for compilation of data at continental scales because of the massive amounts of data produced by their finer resolution.

The 1990s have brought a shift in research efforts from quantifying the extent of deforestation to examining the processes of regrowth following forest clearing. Because of concern with the entire Basin, a number of specialists have relied on single band Thematic Mapper black-and-white photographic products (e.g., use of band 5 TM by Skole and Tucker, 1993, p. 1908). This permits assessment of the total area deforested, and permits refinements such as the potential area affected biologically by deforestation, by taking into account edge effects and fragmentation of forest.

The use of single band TM data has made it possible to stratify Amazonia on the basis of vegetation types, using very general categories that permitted comparison with other studies. Geographic Information System (GIS) techniques have provided a management tool for large amounts of spatial data and a way to merge and geocode information from the more than 200 Landsat scenes encompassing the Amazon (Skole and Tucker, 1993, p. 1906). Skole and Tucker's data for 1988 were compared to single channel MSS data for 1978 using approximately 50 scenes to arrive at changes in forest cover. Skole and Tucker consider the use of single band (i.e., band 5, mid-infrared or 1.55 to 1.75 μm) black-and-white photographic products sufficient to determine deforestation.

Their results agree broadly with earlier assessments using AVHRR—i.e., deforestation was concentrated in a crescent along the southern and eastern

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[2] Landsat TM was launched in 1984 and has 7 bands that capture both the visible and invisible range of the light spectrum. Each band, except the thermal band, has a spatial resolution of 30 m. The thermal band has a resolution of 120 m.

fringe of the Amazon and along major roads and rivers going to the interior. However, single band TM analysis suggests that deforestation estimates based on coarse-resolution satellites such as AVHRR have overestimated deforestation by about 50% (Skole and Tucker, 1993, p. 1909). Towards the end of their article, Skole and Tucker conclude that large areas of regrowth exist throughout the Basin, but they were unable to quantify its extent because of the limitations presented by single band data for differentiating secondary growth from primary forests. One of the reasons for the concern of scientists and policy-makers with deforestation in the Amazon is based on the likely emissions of vast quantities of carbon dioxide from tropical deforestation. Should the regrowth be widespread, it has considerable significance to carbon estimates since growing vegetation is a 'carbon sink'—i.e., it fixes carbon from the atmosphere.

3. Current research on Amazonian regrowth

The recent article by Skole and Tucker (1993) provides a backdrop for the finer-grained analysis that our research team has undertaken for the past 5 years. Our work was designed to link fine-grained analysis to past work which used coarser scales (but continent-wide coverage) and to focus on the possible role of successional processes in counteracting the emission rates of carbon. Sader et al. (1989) were unable to detect tropical forest successional age classes and total biomass differences using the Normalized Difference Vegetation Index [3] (NDVI) and TM data in the Luquillo Forest in Puerto Rico, in part due to the mountainous terrain, the low sun angle, and the shadows cast on steep north and west-facing slopes which reduced spectral reflectance values. They conclude that NDVI from TM may be appropriate to low relief tropical forests but should be used with caution in mountainous regions [4] (Sader et al., 1989, p. 155). Our team has been successful in defining spectral signature patterns for three age classes of secondary growth in four distinct regions of eastern Amazonia (Mausel et al., 1993; Moran, 1993; Brondizio et al., 1994; Moran et al., 1994). These results offer new possibilities to study the role of secondary succession on deforested landscapes, regrowth dynamics, and the temporal effects of land-use systems upon reforestation.

3.1. Methods

Our work began with developing a preliminary classification of vegetation types in two study areas in eastern Amazonia, one in the estuary at Marajó island, the other in an interfluvial moist forest region near Altamira along the Transamazon Highway (Moran, 1981). Research findings from the estuary site may be found elsewhere (Brondizio et al., 1994, 1996; Moran et al., 1994). At Altamira one finds sizable patches of alfisols of medium to high fertility, along with the proverbial nutrient-poor oxisols. The Altamira site is dominated by liana moist forest, but grades to areas of tropical moist forest. The area has an average rainfall of 2000 mm, with a marked dry season of 4 months with 2 months having monthly mean precipitation of only 45 mm. The city of Altamira is located on the Great Bend of the Lower Xingu river. The city had a population of 2000 in 1970, of 11 000 in 1973, and by 1990 reached a population of 80 000 (IBGE, Altamira office statistics).

We used Landsat Thematic Mapper digital data because it had 7 bands from the visible to the thermal infrared (0.45 to 12.5 μm), adequate spatial resolution to permit analysis down to one-hectare fields, and the availability of data for a number of years that permitted the study of short-term (year-to-year) and decade-long dynamics. We felt that the better spatial resolution and broader spectral capabil-

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[3] NDVI is often used at continental scale to analyze AVHRR channel 1 and 2 data to detect green vegetation and arrive at a 'greenness' index. Investigators have related NDVI to seasonal dynamics, tropical forest clearance, biomass estimation, percent ground cover, and other vegetation phenomena.

[4] This is because satellite observations can be influenced by incident solar radiation, radiometric response characteristics of the sensor in the satellite, atmospheric effects, and off-nadir viewing effects (which change the size of the ground resolution cells). This effect is multipied many times over in mountainous terrain.

ities of TM digital data would permit us to make a distinct contribution to the existing literature on deforestation, and we hoped it would allow sufficient discrimination to monitor secondary growth. Up to the present time we have worked with three dates (1985, 1988, and 1991), all of them representing the dry season when cloud cover is considerably less. Furthermore, the scenes are all from a 3-week period that reduces data variability due to seasonality.

Work began from the top-down—i.e., by taking 6 of the 7 channels[5] of digital data and guiding the computer to work on spectral differentiation, supplemented with previous knowledge from the study areas, based on the first author's long-term residence and research in the Altamira area (Moran, 1981). Rather than working initially with whole scenes, subareas of 500 by 500 pixels were analyzed to develop familiarity with spectral patterns using sample areas representing different kinds of land cover, such as untouched mature forest far removed from settled areas to land highly impacted by cattle ranching, roads, and urban development. This is the first step towards linking regional scale analysis with micro-regional land-use patterns. These subareas make it possible to 'think' more locally while dealing simultaneously with regional-scale data, thus providing a preparatory step towards fieldwork. These subareas were subjected to unsupervised classification procedures (cluster analysis) and to multispectral image interpretation. Bands 2, 3, 4, and 5 proved capable of making the best discriminations and clustering of up to 50 classes resulted. The selection of these bands was based on separability analysis. Analysis of the spectral statistics facilitated the reduction of these classes to approximately 18, which were taken as the basis for field studies and ground truthing.[6]

Selection of areas for detailed interviews and soil/vegetation sampling was based on the unsupervised classification developed based upon analysis of spectral statistics. Spectrally-based classes were developed by implementing isodata clustering using Multispec. The cluster statistics associated with a known feature formed the basis for developing core spectral patterns. A Gaussian maximum likelihood classifier algorithm was used in classification in these initial stages. The spectral digital numbers (DN) were interpreted from a theoretical spectral interaction context to provide an initial model of spectral responses. For example, the spectral pattern of water and wetlands differs from other classes because of the significantly lower DN in the near-infrared and mid-infrared bands resulting from the absorption of those wavelengths by water. Dense-green crops with their high chlorophyll content and high moisture content are differentiated from other classes by their high green/red ratio, high near-infrared values, and moderate mid-infrared values. Full details of these spectral analysis decisions are found in Mausel et al. (1993).

In the field we implemented a procedure aimed at ensuring that spatial sampling bias was not unwittingly introduced and accurate geocorrection occurred. Twenty large subsets were marked on the scene, spread over the entire region, so that all parts of the image were visited and represented in the analysis. In each area visited, precise locations (latitude and longitude) were obtained using a global positioning system (GPS) device.[7] Field observations were made of classes procured through unsupervised classification, emphasizing forest and secondary successional age classes.

Throughout the area of interest, interviews were conducted with farmers in order to obtain land use histories. The principal author visited the farms which appeared to provide secondary successional sites for sampling based upon the Landsat TM spectral analysis previously carried out in the lab. Candidates were selected in an effort to distribute vegetation classes throughout the classified image. To reduce possible error in locating study areas, candidate sites that

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[5] We did not use channel 6, the thermal channel, because it has coarser resolution (120 m rather than 30 m) and would have degraded the spatial resolution of the analysis.

[6] "Ground truthing" is a term used for the process of checking in the field the accuracy of what has been deduced from the spectral statistics in a satellite image. It is also a way to improve the accuracy of supervised classification. Our approach goes far beyond the usual approach to ground truth of simply verifying the lab analysis, by actually using the detailed field information to change classes and understand the dynamics of land cover change.

[7] We used a Magellan 1000 Pro device at the time of the 1992 and 1993 field studies.

were larger in size were generally preferred over very small ones which could be a product of error in these early stages of image classification, or could prove ephemeral through time.[8] Upon arrival, a discussion with the farmer took place, followed by a visit to the forest, secondary successional area, or pasture of interest. While observing the area, one could verify its history and age—and ensure its continued existence to the present—followed by a detailed interview about household demographic composition, range of economic activities of the household, and a detailed history of the land use on the property, particularly the area to be sampled. In some cases it was discovered that the area no longer existed in the same state that was true for the satellite data available (i.e., that the area had been altered since the date of the satellite image), which made it a poor candidate for soil/vegetation sampling. While such a property might not be sampled in depth, general observations of land use were made and the information obtained from the interview were still of value in order to further spectral analysis and supervised classification. If the property was appropriate for sampling, the principal author asked permission from the property owner to have the team come the next day (or a convenient day and time) to carry out the detailed soil/vegetation inventory.

From these interviews we selected sites of differently aged classes of secondary growth that would become the focus of field sampling. With a crew of 6 collaborators, 22 plots were sampled in 1992 at the Altamira site and an additional 14 samples were added during fieldwork in 1993 for a total of 36 soil and vegetation inventories. A few of the 1992 sampled areas could not be used in this paper because of experimentation with plot sizes during the first year. Plots from 1992 which differed from the standard size in 1993 were excluded for the purpose of statistical analysis. At each sampled area, we took soil profile samples to 1 m depth, counted and identified species, percentage of covered ground, and measured stem height and total height of plants if equal to or over 10 cm diameter at breast height and/or over 2 m in height.

The field-based research, supported by the previous laboratory land-use analysis, represents a 'bottom-up' approach that characterizes our research methodology and provides a contrast to other remote sensing analyses in Amazonia. While it is a usual practice among remote sensing analysts to carry fieldwork only as a procedure to check unsupervised classifications, our study assigns fieldwork a very different role. As new information concerning vegetation classes and dynamics emerged from fieldwork, we used it to re-interpret and change previous classifications and even to create new classes of land cover. Unsupervised spectral analysis and simple verification of already created classes can obscure the realities of a complex landscape. The use of field data, especially vegetation information and land-use histories, gives new meaning to spectral statistical patterns and brings to light local differences in the process of land cover change within a large region.

3.2. Regional scale vegetation change

Areas studied included major vegetation types found in the region: dense moist and liana forests, pastures, pastures invaded with palms such as inaja (*Maximiliana maripa*, now *Attalea maripa*) and babaçú (*Orbignya phalerata*, now *Attalea speciosa*) (Henderson, 1995), and fallows representing age classes 0–2, 3–5, 6–10, 11–15, 16–20 and over 20 years. After 1992 we aggregated 0–2 and 3–5 data into a 0–5 interval of early secondary succession (or SS1) due to the difficulty in attaining high degrees of accuracy in the spectral analysis between 0–2 and 3–5. When aggregated into this 5-year interval, accuracy reached 88.3% (Mausel et al., 1993). Cropland and agroforestry plantations were also included, but will be the focus of a separate paper.

At regional scale, using a combination of spectral analysis and field research, it is possible to observe in Altamira a decline in the area of mature forest from 64.01% in 1985 to 58.73% in 1988 to 56.99% in 1991 (see Table 1). Pastures declined in areal extent during this same period. Relatively homoge-

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[8] Another reason to choose larger areas is that due to Selective Availability in GPS technology, as much as a 100-m error is possible in location. This is a policy of the Defense Department in the U.S. applied to civilian GPS receivers. This error can be reduced through differential correction; nevertheless, chances for error increase if the study area is smaller than 3 pixels or 0.9 hectares.

Table 1
Land cover changes in the Altamira study

Land cover class	Percent 1985	Land 1988	Cover 1991
Mature forest	64.01	58.73	56.99
Pasture	10.82	7.92	3.06
Initial SS (SS1)	5.58	8.58	10.9
Intermediate SS (SS2)	4.19	9.97	15.47
Advanced SS (SS3)	0.91	3.96	5.93
Bare	7.59	1.71	1.33
Crop	1.16	2.82	0.64
Water	4.87	5.35	5.29
Wetland	0.87	0.96	0.39

[*] These statistics are derived from classification of the 267 000 ha study area.

Source: Mausel et al. (1993).

nous pastures declined from 10.8% of the area in 1985 to 7.9% in 1988 to 3.06% in 1991. The most notable increases came in secondary successional vegetation: early secondary succession, 0–5 years (hereafter SS1), often represented by the typical degraded pasture, went from 5.58% in 1985 to 8.58% in 1988 to 10.9% in 1991. Even more impressive is the increase in the 6–10 year age-class of secondary succession (hereafter SS2) which rose from 4.19% in 1985 to 9.97% in 1988 to 15.47% in 1991. At this stage many householders clear 6–10 year regrowth and establish either a clean pasture or cropland. However, this tends to occur more frequently if the property is near markets and a good road, if the price of commodities is favorable, and if credit is available to pay the labor costs required for land preparation. When those conditions are absent, the vegetation is likely to evolve into the third stage of secondary growth, advanced secondary succession or regrowth of over 11 years (hereafter SS3). While SS3 accounted for less than 1% in 1985, it had increased to 3.96% by 1988 and to 5.93% by 1991. These changes are the expression of different developmental processes taking place in the region since the 1970s. The most impressive change is the failure to implement the intended cash crop economy among small farmers. Instead, one sees the failure of pasture formation, despite all the fiscal incentives and economic advantages it received in the 1970s and early 1980s. At regional scale what one sees is a process of secondary successional forest management based on fallows of about 10 years, with a small proportion being abandoned for longer, and some shorter, periods.

3.3. Vegetation structure and composition

Vegetation structure is an important component of ecological and land-use analysis. Ecologically, it reflects the pace of regrowth and accumulation of biomass which can be used as a good indicator of sustainability of land-use. In terms of land use, vegetation structure is a basic parameter to classify vegetation types, and is the main component of spectral analysis of satellite data. To understand regrowth stages based on structural classes can give support to improve analysis of satellite-derived data and, most importantly, provides parameters to understand site restoration and potential use of fallows through enrichment of the sites with valuable hardwood species.

Analysis of the vegetation structure and composition supports the spectral analysis and classification of land cover by means of satellite digital data. The differences in structural characteristics (i.e., total height, stem height, and basal area) between the classes developed are all unlikely to have occurred by chance (ANOVAS, $P \leq 0.15$). This level of confidence confirms our decision to group by the categories developed in the initial spectral analyses.

The composition of invading species is dependent on the particularities of previous land use, composition of surrounding vegetation, competitive interactions, and other factors such as the seed bank and local topography. Fallows that are weeded infrequently tend to have a greater presence of woody pioneer species, in contrast to dominance of grasses and herbs found in frequently weeded areas (Uhl et al., 1982). Unlike older fallows, where species that occur at the highest densities are commonly the most frequent, this is not true during the first 5 years of succession. Grasses and herbs, such as *Desmodium canum*, *Elephantopus mollis* and *Acalypha arvensis*, are spatially clumped at high densities. Palms and pioneer shrubs, such as *Lantana camara*, *Wulffia baccata*, *Cyperus flavida* and *Orbygnia phalerata*, were the most evenly distributed species in initial fallow stages. By the third year of this stage, up to 88 species (trees, shrubs, and herbaceous) were found at our sampled sites.

Table 2

Summary of means and standard errors for variables with sites grouped by successional age (dbh > 10 cm)

Successional stage	$n =$	Measured variables					
		Total height (m)	Stem height (m)	Species richness (species no./1500 m^2)	Diversity (Shannon's index)	Density (stems/ha)	Basal area (m^2/ha)
SS2	3	11.84 ± 0.88	7.17 ± 0.65	25	2.41 ± 0.13	711.11 ± 66.93	15.52 ± 0.94
SS3E [1]	4 *	11.72 ± 0.84	6.01 ± 0.72	29	2.63 ± 0.22	803.33 ± 114.70	21.79 ± 5.15
SS3L [2]	4	16.17 ± 2.11	10.68 ± 2.07	32	2.78 ± 0.42	663.75 ± 113.43	21.19 ± 4.61
Mature forest	2	15.16 ± 0.03	9.24 ± 1.49	52 **	3.48 ± 0.18	695 ± 20.00	38.76 ± 9.00

[1] E = fallows of 11–15 years. [2] L = fallows of 16 years and older.

* For basal area in this class, $n = 3$ due to problems with data collection.

** This measure is species no./2500 m^2.

In the next fallow stage, SS2 or regrowth of 6–10 years, increased canopy height causes shading which is likely to be a factor in the growth and continued presence of pioneer species. Mean height increases from 3.75 m in SS1 to 11.84 m in SS2. *Cecropia* and to a lesser extent *Orbygnia* are commonly encountered and tend to have higher importance values [9] at this stage. These species are frequently indicators of human disturbance (Henderson, 1995).

In the advanced stages of secondary succession, SS3E, there is an increasing number of individuals with greater total height. Mean height is slightly less than in SS2 due to the high mortality of individuals in the genus *Cecropia* which were such a large part of the SS2 population in this area (the process may differ elsewhere—e.g., if *Cecropia* is not a dominant in SS2). This fast-growing species is largely responsible for the high mean height of SS2 vegetation, as well as for the lower mean height for SS3 when they begin to senesce and experience high mortality. Moreover, the increased importance of mature forest species, such as *Neea floribunda*, *Cenostigma tocantinum* and *Bertholletia excelsa*, is apparent in this successional stage.

Beyond 16 years it is sometimes structurally as well as spectrally difficult to distinguish mature forest from secondary growth of advanced age (Li et al., 1994). Grasses and shrubs are no longer commonly found. Basal area is virtually the same in our SS3 fallows. There is greater tree species diversity in this late stage. Table 2 summarizes the means and standard errors for each of these classes' density, species richness, diversity, height and basal area.

How long it takes for these growing fallows to return to species equivalency to mature forest is very difficult to determine. Generalizations that suggest 100, 300 and 500 years are based on reasonable extrapolations, but the range of these predictions is too wide and too site-specific to be very helpful. In mature forest the mean density of stems per hectare is relatively low while the basal area is almost double that of secondary growth of SS3L. A few very large individuals contribute the bulk of this increase in basal area. This also means that while it is common to find species with height over 30 m, mean tree canopy height may be only slightly higher than 15 m in these mature liana forests due to the presence of understory species.

The average total number of families is highest in SS1 and Forest (i.e., 44 families) while SS2 and SS3 reveal a slightly lower average (see Table 3). However, when one considers families of trees ≥ 10 cm dbh, a slow steady increase in the number of families appears as the vegetation progresses from a 7-year fallow to forest which may have as many as 28 families of trees (see Table 4). The high number of families in SS1 may be explained by the wider spectrum of plant types—grasses, herbs, pioneer species, etc.—colonizing the area. In addition to the increase with age of families with trees ≥ 10 cm dbh, there is a corresponding shift in family dominance. Forest exhibits clear dominance of the family Caesalpiniaceae, while SS2 shows dominance of the family Mimosaceae and Cecropiaceae (see Table 5).

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[9] Importance value is the average of relative density, relative frequency, and relative basal area summed.

Table 3
Diversity of families

Age	Plot	Total no. of families	Average total no. of families
SS1			
3	009-93	46	44
3	010-93	43	
3	004-93	44	
SS2			
7–8	014-93	45	38
8	007-93	32	
±8	006-93	38	
SS3E			
10	005-93	43	38
10–11	011-93	33	
10–11	012-93	35	
10–11	013-93	39	
SS3L			
16	001-92	26	34 *
15–17	001-93	41	
15–17	002-93	37	
17	005-92	32	
Forest			
Mature	003-93	41	44
Mature	008-93	47	

* Without the outlier of 26 families in plot 001-92, the average number of families is 37, showing little variation from early SS3 and SS2.

Soils also seem to play some part in family diversity. The trend in the data appears to be that in SS2, the nutrient-rich alfisols support a greater number of families than do oxisols. When comparing within similar soil conditions, plots that have been used for pasture as well as agriculture support a greater number of species than those only used for crops—a surprising and unexpected finding. It must be considered that this higher number may represent largely an increase in pioneer species, which will die off later, thereby reducing total number of species.

An increase in the percentage of mature forest species is apparent with time [10] (Table 6). The greatest increase in our sampled plots takes place between SS1 and SS2, and between SS2 and SS3E, with a 10% increase in each. The rate of increase in the number of mature forest species slows upon reaching SS3E, suggesting that by this stage of regrowth the majority of common mature forest species have made their way into the fallow area for the next several decades. Seeds of rare species will reach these older fallows only slowly [11] and a return to original species diversity may be very slow indeed.

Three factors were examined for the degree to which they exercised control over vegetation characteristics. None of the three factors (age, land use and soil type) shows significant effect on tree density—which appears to be dependent upon which species are able to colonize the area first. Differences in total height and stem height appear to be correlated with land use differences at a statistically significant level (ANOVAS, $P \leq 0.001$). Alfisols supported a higher basal area than oxisols ($t = 2.586$, $P \leq 0.005$). No other variables were significantly different between soil types.

In short, it can be said that land use has a significant effect on the variables total height, stem height, and basal area. Age alone accounts for 58% of the variance seen in total height, while land use alone accounts for 82% of the variance seen in total height (W^2bet) (Lindman, 1992). Both of these factors play an important role in determining the differences in total height. Land use clearly accounts for a greater proportion of the variance than other variables. The Shannon-Weiner diversity index is the only variable for which neither land use nor age accounts for a significant amount of the variance. It is most likely that the vegetation surrounding the fallow, as well as the size of the area plays a large determinant role on the area's species diversity. Distance from mature forest and the presence of contiguous stretches of older successional vegetation between the abandoned fallow and mature forest are two factors that have been shown to be determinants of species composition elsewhere (Uhl et al., 1990).

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[10] As a baseline of mature forest species we compared our plots to the inventory of Mario Dantas (1988) in the Altamira Region.

[11] It needs to be pointed out that the number of species is very sensitive to the size of the area sampled. SS1 sampled areas were 100 sq.m, whereas SS3 and Forest were 1500 sq.m, an increase necessary to capture the growing dominance of trees.

Species diversity is a community attribute reflecting the influence of seed dispersal, competition, predation and local extinction (Peet, 1992 in Glenn-Lewin et al., 1992, p. 136). A variety of patterns has been reported in the literature as a result of differences in the environmental context. It has been suggested that on favorable sites diversity quickly increases to a high level, but then declines dramatically during the thinning out phase (Peet, 1978, 1988). But this pattern is not always found. In the study area of Altamira, where patches of both nutrient-rich and nutrient-poor soil occur, and with a broad array of land use patterns, it is best to interpret species diversity during succession as a result of the combined influence of the form of land clearing procedure used, the length of time the land was cultivated, the initial soil fertility at the site, the intensity of cultivation or use and the species interactions at the particular site.

4. Conclusions

Wessman (1992, p. 180) has called for studies that link ground observations to regional and global scales if we are to take full advantage of the detailed data available at different scales. A number of these research efforts are currently ongoing, but they have paid scant attention to the human dimensions of these processes. Extrapolation of microecosystem research to regional and global scale has been hindered in the past by difficulties in observing large-scale spatial heterogeneity and long-term patterns of successional dynamics. Remote sensing linked to ground-based successional studies provides the most promising of tools for understanding ecosystem structure, function, and change. The capacity to detect long-term change in ecosystems can be enhanced by analysis of image texture combined with spatial statistics that permit analysis of stand struc-

Table 4
Plot characteristics

Age (years)	Plot No.	No. of families	No. of species/area sampled	No. of individuals/ha	Shannon-Weiner diversity index
SS1					
3	009-93	46	88	173700	3.278
3	010-93	43	73	80200	3.406
3	004-93	43	68	98400	3.125
SS2		> 10 cm dbh	> 10 cm dbh	> 10 cm dbh	> 10 cm dbh
7–8	014-93	17	27	773	2.136
±8	006-93	15	26	753	2.651
8	007-93	13	23	580	2.362
SS3E					
10	005-93	23	33	553	3.033
10–11	011-93	17	34	880	2.783
10–11	012-93	14	22	1080	2
10–11	013-93	17	25	707	2.696
SS3L					
15–17	001-93	25	41	560	3.429
15–17	002-93	26	43	493	3.541
16	001-92	12	20	613	2.31
17	005-92	13	23	995	1.824
Forest					
Mature	003-93	26	52	675	3.654
Mature	008-93	28	52	715	3.299

[*] SS1 samples all individuals within a 100 m^2 area while the later stages include all individuals 10 cm dbh and above within a 1500 m^2 area or 2500 m^2 area, in the case of mature forest.

Table 5
Family composition *

Age (years)	Plot No.	Family with highest no. of species	Family with highest no. of individuals	Family of most important ** species
SS1				
3	004-93	Poaceae (5)	Fabaceae (196)	Fabaceae—*Desmodium canum* (0.11767)
3	009-93	Asteraceae [1] (5)	Fabaceae (339)	Fabaceae—*Desmodium canum* (0.10123)
3	010-93	Aseraceae [2] (6)	Verbenaceae (142)	Verbenaceae—*Lantana camara* (0.10370)
SS2				
7–8	014-93	Mimosaceae (4)	Cecropiaceae (66)	Cecropiaceae—*Cecropia obtusa* (0.23574)
±8	006-93	Mimosaceae (4)	Cecropiaceae (43)	Cecropiaceae—*Cecropia palmata* (0.30805)
8	007-93	Mimosaceae (4)	Cecropiaceae (41)	Cecropiaceae—*Cecropia palmata* (0.46908)
SS3E				
10	005-93	Caesalpiniaceae [3] (3)	Caesalpinaceae (17)	Caesalpinaceae—*Cenostigma tocantinum* (0.28727)
10–11	011-93	Mimosaceae (7)	Mimosaceae (32)	Tililaceae—*Apeiba burchellii* (0.18143)
10–11	012-93	Arecaceae [11] (3)	Arecaceae (85)	Aracaceae—*Orbignya phalerata* (0.53149)
10–11	013-93	Arecaceae [3] (3)	Cecropiaceae (23)	Cecropiaceae—*Cecropia palmata* (0.24521)
SS3L				
15–17	001-93	Mimosaceae (6)	Euphorbiaceae (13)	Euphorbiaceae—*Sagotia racemosa* (0.18936)
15–17	002-93	Caesalpinaceae [22] (4)	Myristicaceae (8)	Myristicaceae—*Virola melinonii* (0.17799)
16	001-92	Mimosaceae (4)	Cecropiaceae (59)	Cecropiaceae—*Cecropia obtusa* (0.41952)
17	005-92	Mimosaceae (5)	Euphorbiaceae (88)	Euphorbiaceae—*Sapium marmieri* (0.38519)
Forest				
Mature	003-93	Caesalpinaceae (6)	Caesalpinaceae [33] (16)	Fabaceae—*Alexa grandiflora* (0.2558)
Mature	008-93	Caesalpinaceae (9)	Caesalpinaceae (43)	Caesalpinaceae—*Cenostigma tocantinum* (0.30201)

[1] Also Euphorbiaceae, Mimosaceae and Rubiaceae; [2] also Euphorbiaceae; [3] also Mimosaceae; [11] also Mimosaceae and Moraceae; [22] also Fabaceae; [33] also Sterculiaceae.

* Only families of individuals of 10 cm dbh and above are counted for SS2 and the later stages.

** The most important species implies highest importance value at the site. The actual value is given in parentheses

ture from satellite data (Wessman, 1992, p. 189). For a Northeastern boreal forest, a 10-year time series Landsat data set was used to track changes in succession stages. Once the images were rectified for changes in atmospheric conditions between years, it was possible to infer the dynamics taking place.

Table 6
Percentage of mature forest species found in each study site

Land use	SS1	SS1	SS1	SS1	SS1	SS1	SS2	SS2	SS2	SS3E [1]	SS3E	SS3E	SS3E	SS3L [2]	SS3L	SS3L	SS3L
Cleared/ burned only														30	26.1		
Crop	8.5	10.9					26.1	22.2		28						36.6	32.6
Pasture			17.2	5.4													
Crop and pasture					9.5	3.5			11.5		36.4	23.5	31.8				
Age	Mean Percent																
SS1	9.2																
SS2	19.9																
SS3E [1]	30																
SS3L [2]	31.3																

[1] E = fallows of 11–15 years. [2] L = fallows of 16 years and older.

Similar procedures, with the added advantage of higher-resolution TM data, have been implemented in our Amazonian study with results that permit assessment of successional processes from spectral analysis. We have further tested and verified their linkage to differences in soil fertility, land use history, and time since abandonment.

Sustainable development requires strategies for renewal of resources. As ecosystem restoration begins to take place in parts of Amazonia, it will be necessary to have baseline data on successional pathways following a range of disturbances if these pathways are to be manipulated (Luken, 1990, p. 19). We have learned that Amazonian forests are more structurally resilient than fragile, that the magnitude of the disturbance and its temporality are directly related to the restoration rate. We have also learned that the declining rates of deforestation between 1988 and 1993 were largely due to economic stagnation and hyperinflation besetting Brazil—and that more proactive policies than recession are needed to reduce the high rates of new deforestation. In an area as vast as Amazonia, understanding these processes cannot rely on field surveys or satellite data alone. Instead, a combination of nested and differentially scaled data collection is necessary. The approaches discussed here provide one such opportunity to generate fundamental data to inform ecosystem restoration, improve carbon modeling accounting, and a balance between use and conservation of this rich realm of nature.

From the spectral analysis of the satellite data we have learned the spatial distribution of deforestation, its regional scale, and its areal extent. We have discovered its dynamic nature through time. One can see a greater frequency of persistent pastures near towns, where other sources of income subsidize the costs of fighting off the invasion of woody regrowth, and the greater frequency of degraded pastures and abandonment of previously cleared land with increased distance from town and the main trunk of the highway. Careful examination of the spectral data aided in the subsequent collection of detailed household surveys and soil/vegetation surveys. These in turn provided insight into the behavior of households and the outcome of their decisions on the landscape.

This information leads us to conclude that both age and land use have important effects on the characteristics and rate of regrowth following crop or pasture abandonment. The type of land use which is least detrimental in the long run will depend upon which traits are judged to be most important. Diversity seems to be higher in plots that have been used for both agriculture and pasture before being abandoned than in those that were only cropped before abandonment. This is an unexpected finding, but it has recently been confirmed at our other research sites, and merits continued research and study to capture its implications. Whether it means that one finds more invasive species in previously pastured areas, or whether it favors particular kinds of woody and vine species, is an open question. It could also be the case that species distributions are more even. Early successional communities are typically controlled by density-independent regulators; thus, competition may not be sufficiently strong as to create dominance of few species. Diversity is lower still in control plots that were burned and abandoned but never planted. These control plots, cut-burned but never planted, somewhat resemble the experience of selectively logged areas experiencing spontaneous fire. This lower diversity finding has considerable implications in terms of the future impact on biodiversity of selective logging and its association with increased proneness to fire. Such widespread fires in selectively logged areas could have a far more negative impact on biodiversity than cutting–burning–planting and establishment of pastures. Nutrient-rich soils, such as alfisols, seem to promote faster growth rates as expressed in greater basal area. A great deal more research is necessary to fully understand what impact local forms of land use have on the species diversity and total biomass of regrowing vegetation in sites with different initial conditions of fertility, drainage, and proximity to markets, to name but a few of the possibly relevant factors that need to be accounted for.

Following initial deforestation, farmers are confronted with ever-increasing labor and capital costs in controlling woody regrowth. Over time, the rate of deforestation of mature forest by households declines as more of their effort is spent on re-using land previously cleared and in some stage of secondary succession. Understanding how to achieve a sustainable land use strategy that meets household needs while maintaining viable tracts of mature forest will become a matter of growing interest to communities in the Amazon, to regional and national policy makers, and to all those who are concerned with the consequences of deforestation.

It is clear that farmers in Altamira, and elsewhere, use secondary successional forests, thereby reducing their demand for deforestation of primary forests. They use these secondary forests not only for crops and pastures, but also for agroforestry plantations, including hardwood plantations and fruit tree plantations that are species-diverse and economically valuable. Determining what proportion of the fires burning in the Amazon since 1995 are derived from primary forests, and how much of it is re-use of previously deforested land, is a matter of considerable significance. The tools are now available to answer these questions.

Acknowledgements

This paper would not have been possible without the collaboration of many team members from a variety of disciplines, among them: Amy Gras from Indiana University; Paul Mausel and You Wu from Indiana State University at Terre Haute; many colleagues from EMBRAPA in Brazil, among them Mario Dantas, Lucival Rodrigues Marinho, Jair da Costa Freitas, Italo Claudio Falesi, and Adilson Serrao, who provided field assistance and/or institutional support. We appreciate the critical suggestions made by Robert Walker and the journal reviewers. This work was supported by grants from the National Science Foundation (SBR 9100526 and SBR 9310049) and by the Midwestern Center of the National Institute for Global Environmental Change and is gratefully acknowledged. None of these agencies should be held responsible for the views presented herein.

References

Anderson, A., 1990. Deforestation in Amazonia: dynamics, causes and alternatives. In: A. Anderson (Editor), Alternatives to Deforestation: Steps toward Sustainable Use of the Amazon Rain Forest. Columbia University Press, New York, NY, pp. 3–23.

Booth, W., 1989. Monitoring the fate of forests from space. Science, 243: 1428–1429.

Brondizio, E., Moran, E., Mausel, P. and Wu, Y., 1996. Land cover in the Amazon Estuary: linking of the thematic mapper with botanical and historical data. Photogram. Eng. Remote Sensing, 62: in press.

Brondizio, E., Moran, E., Mausel, P. and Wu, Y., 1994. Land use change in the Amazon Estuary: patterns of Caboclo settlement and landscape management. Hum. Ecol., 22(3): 249–278.

Dantas, M., 1988. Studies on Succession in Cleared Areas of Amazonian Rain Forest. Ph.D. Dissertation, Oxford University, Oxford.

Glenn-Lewin, D., Peet, R. and Veblen, T., 1992. Plant Succession: Theory and Prediction. Population and Community Biology Series No. 11, Chapman and Hall, London.

Henderson, A., 1995. Palms of the Amazon. Oxford University Press, Oxford, UK.

Li, Y., Mausel, P., Moran, E. and Brondizio, E., 1994. Discrimination between advanced secondary succession and mature forest near Altamira, Brazil, using Landsat TM data. In: Proceedings of the American Society for Photogrammetry and Remote Sensing, pp. 350–364.

Lindman, H.R., 1992. Analysis of Variance in Experimental Design. Springer-Verlag, New York, NY.

Lugo, A. and Brown, S., 1993. Management of tropical soils as sinks on sources of atmospheric carbon. Plant Soil, 149(1): 27–41.

Luken, J.O., 1990. Directing Ecological Succession. Chapman and Hall, London.

Mahar, D., 1988. Government Policies and Deforestation in Brazil's Amazon Region. World Bank, Washington, DC.

Matson, M. and Holben, B., 1987. Satellite detection of tropical burning in Brazil. Int. J. Remote Sensing, 8(3): 509–516.

Mausel, P., Wu, Y., Li, Y., Moran, E. and Brondizio, E., 1993. Spectral identification of successional stages following deforestation in the Amazon. Geocarto Int., 8(4): 61–71.

Moran, E.F., 1981. Developing the Amazon. Indiana University Press, Bloomington, IN.

Moran, E.F., 1993. Deforestation and land use in the Brazilian Amazon. Hum. Ecol., 21: 1–21.

Moran, E., Brondizio, E., Mausel, P. and Wu, Y. 1994. Integrating Amazonian vegetation, land use and satellite data. BioScience, 44(5): 329–338.

Nepstad, D., Uhl, C. and Serrao, E.A.S., 1991. Recuperation of a degraded Amazonian landscape: forest recovery and agricultural restoration. Ambio, 20(6): 248–255.

Peet, R., 1978. Forest vegetation of the Colorado Front Range: patterns of species diversity. Vegetatio, 37: 65–78.

Peet, R., 1988. Forests of the Rocky Mountains. In: M. Barbour and W. Billings (Editors), North American Terrestrial Vegetation. Cambridge University Press, Cambridge, pp. 63–101.

Peet, R., 1992. Community structure and ecosystem function. In: D.C. Glenn-Lewin, R. Peet and T. Veblen (Editors), Plant Succession: Theory and Prediction. Chapman and Hall. London, pp. 103–151.

Sader, S., Stone, T. and Joyce, A., 1990. Remote sensing of tropical forests: an overview of research and applications using non-photographic sensors. Photogram. Eng. Remote Sensing, 56(10): 1343–1351.

Sader, S., Waide, R. Lawrence, W. and Joyce, A., 1989. Tropical forest biomass and successional age relationships to a vegetation index derived from Landsat TM data. Remote Sensing Environ., 28: 143–156.

Setzer, A. and Pereira, M.C., 1991. Amazonia biomass burnings in 1987 and an estimate of their tropospheric emissions. Ambio, 20: 19–22.

Skole, D. and Tucker, C., 1993. Tropical deforestation and habitat fragmentation in the Amazon: satellite data from 1978 to 1988. Science, 260: 1905–1910.

Stone, T., 1994. Analysis of selective logging using Landsat TM data of E. Brazilian Amazonia. Paper presented at Association of American Geographers, 90th Annual Meeting, 29 March–2 April.

Tucker, C.J., Gatlin, J.A. and Scheider, S.R., 1984. Monitoring vegetation of the Nile Delta with NOAA-6 and NOAA-7 AVHRR imagery. Photogram. Eng. Remote Sensing, 50: 53–61.

Uhl, C., Clark, H. and Maquirino, P., 1982. Successional patterns associated with slash-and-burn agriculture in the upper Rio Negro region of the Amazon Basin. Biotropica, 14(4): 249–254.

Uhl, C., Nepstad, D., Buschbacher, R., Clark, K., Kauffman, B. and Subler, S., 1990. Studies of ecosystem response to natural and anthropogenic disturbances provide guidelines for designing sustainable land-use systems in Amazonia. In: A. Anderson (Editor), Alternatives to Deforestation: Steps toward Sustainable Use of the Amazon Rain Forest. Columbia University Press, New York, NY, pp. 3–23.

Wessman, C., 1992. Spatial scales and global change: bridging the gap from plots to GCM grid cells. Annu. Rev. Ecol. Syst., 23: 175–200.

Woodwell, G., Houghton, R.A., Stone, T. and Park, A., 1986. Changes in the areas of forests in Rondonia, Amazon Basin, measured by satellite imagery. In: J. Trabalka and D. Reichle (Editors), The Changing Carbon Cycle: A Global Analysis. Springer-Verlag, New York, NY, pp. 242–257.

Woodwell, G., Houghton, R., Stone, T., Nelson, R. and Kovalick, W., 1987. Deforestation in the Tropics: new measurements in the Amazon Basin using Landsat and NOAA AVHRR. J. Geophys. Res., 92: 2157–2163.