

AN ABSTRACT OF THE DISSERTATION OF

Scott Bergen for the degree of Doctor of Philosophy in Forest Science presented on August 19, 2004.

Title: Spatial and Temporal Dynamics of Anthropogenically Influenced Forests of the Brazilian Amazon.

Abstract Approved:

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The Amazon Tropical Rain Forest is the largest tropical rain forest system, comprising approximately 65% of such forests on earth. Since the 1960's, human populations in the Brazilian Amazon have increased from two to over 20 million. Concomitant urbanization, forest conversion, and economic dependence on resource extraction have exerted severe demands on these fragile ecosystems. Its status as a global carbon source and sink and its unique biodiversity compels study of the Amazon and the dramatic changes sustained to optimize conservation strategies and ameliorate loss. While the overall effects of deforestation are well documented, little is known about the spatio-temporal interactions between forest dynamics and anthropogenic disturbance. As part of a collaborative NASA-Smithsonian Institute project, the present research focuses on the analysis and modeling of spatio-temporal forest dynamics in the Amazon across scales. The first analysis indicates that secondary forest tree species richness is significantly influenced by proximity to large forest tracts containing frugivorous fauna. These results underscore the necessity of conserving large tracts and the consideration of spatial design for adequate preservation of biodiversity. Methods developed in the second analysis

demonstrate that while deforestation devastates biodiversity, secondary forests may contribute to the attenuation of greenhouse gas concentrations. However, this carbon tracking process is revealed to be more complex than previously assumed. Land tenure dynamics and heterogeneity preclude the use of simple static models. These results imply that modeling the world's largest rain forest's effects on global greenhouse gas concentrations is highly uncertain and must be evaluated cautiously. The final analysis evaluates the diverse effects of specific large-scale development projects on forest integrity currently being planned. Riverside deforestation and road upgrades have the highest potential for effecting irreversible environmental damage. Protected forests and indigenous lands comprise 30% of the Brazilian Legal Amazon and have the potential to prevent the degradation of 464,151 km² of mature forest. However, political and economic pressures on indigenous peoples threaten to destabilize land tenure and land-use practices. Urgent efforts need to be directed to develop long-term and large-scale planning of Amazon land-use that takes such complexity into consideration.

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**Spatial and Temporal Dynamics of Anthropogenically Influenced
Forests of the Brazilian Amazon**

**by
Scott Bergen**

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Scott Bergen, Author

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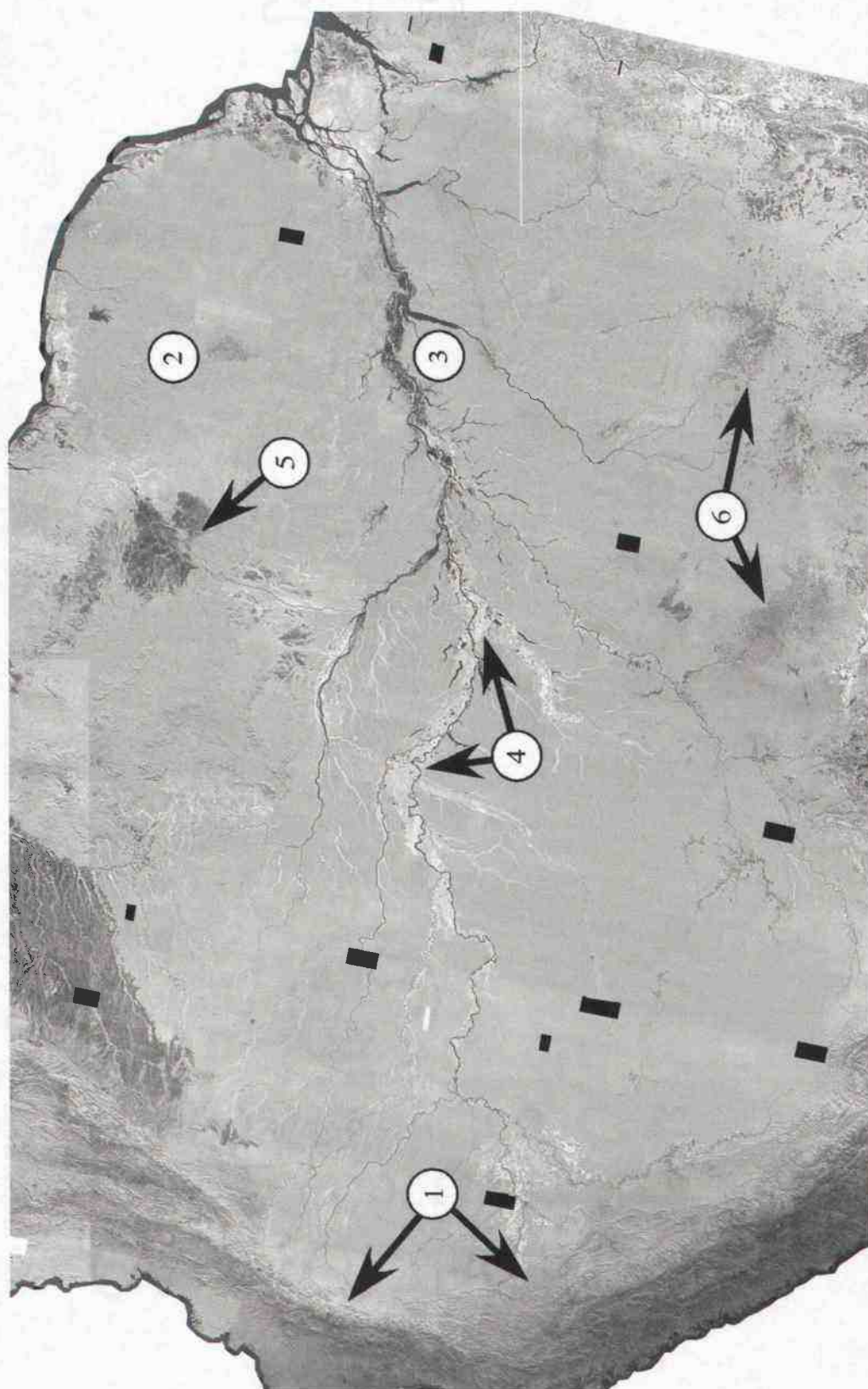
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INTRODUCTION TO AMAZON FORESTS, GIS AND REMOTE SENSING

INTRODUCTION

The Amazon Basin is South America's largest watershed comprising well over 6,000,000 km². The countries of Bolivia, Brazil, Columbia, Ecuador, and Peru contain Amazon watershed forests with adjacent Venezuela, British Guyana, French Guyana and Suriname also containing significant amounts of tropical moist forest. The Amazon basin is characterized by a central low elevation plane with meandering tributaries (Figure 1). These forests exist on soils that have become leached of nutrients due to high rainfall and temperatures. Many of these forests are inundated by seasonal rises in the water level. Inundated 'gallery forests' have little under-story development and are called '*várzea*.' To the west of the central basin the Amazon is confined by the Andean Mountains that are the headwaters for much of the basin. Forested systems exist to an approximate elevation of 1,300m (Bush *et al.* 2004 within a range of 1,300 to 2,000 m) where at higher elevation shorter stature cloud forests attain dominance. These Andean forests are characterized by high beta-diversity with many tree and plant species confined to small edaphic distributions. Forests in proximity to the 'Andean Crescent' have higher nutrient concentrations in their soils due to the geologically recent mountain formation. Amazon forests in the northeast are of a geologic formation called the Guyana Shield. This formation is Precambrian in origin, contains very low soil nutrient availability and is more varied in topography relative to the central basin. Portions of the Guyana Shield do not support forest because of high elevation and/or decreased rainfall. These forests contain diverse forest tree species (285- 301 spp. ha⁻¹) across a great expanse that supports a high faunal species diversity (Olivera and Mori, 1999). This high alpha

Figure 1.1 The Amazon Basin in 1995 as depicted from the Japanese Engineered Radar Satellite (JERS). The figure is an image composite created from the JERS sensor when the portions of the Amazon Basin were at high water. The number 1 shows the western boundary of the Amazon basin called the Andean Crest. Number two is the geologic formation of the Guyana Shield. Number three is the region of the central Amazon plane. Number four shows inundated *várzea* forests at high water. *Várzea* has a whiter appearance with the radar technology used. Number five is naturally occurring savanna. Number six shows areas that have been deforested by non-endemic human factors.



diversity distribution is rarely interrupted by abrupt changes in edaphic or climatic conditions (with the exception of the Guyana Highlands).

The complete species diversity within the Amazon basin remains undocumented and unknown. Recent best guesses of the species diversity contained within the region are between $10\text{--}50 \times 10^6$ species with the vast majority contained within the arthropods (Erwin 1983, Wilson 1988). Even with this vast preponderance of beetles, new genera of trees (Dave Niell *pers comm*) and charismatic primates are being discovered and described at a frequent and constant rate throughout the nineties (Mittermeier 2002, as reported in National Geographic with 13 species of primate being discovered since 1980). The richness of diversity is not only confined to the animals and plants. The Amazon Basin also contains the highest number of human languages relative to any other region of the world (Meggers 1975, Hinchberger 2002). Each language can be thought of as representing an isolated population or deme of human evolution. However, modern colonization including incursion, disease, and exploitation is decimating indigenous cultures (Wallace 2003).

Science has renewed interest in the Amazon Basin beyond its high species diversity. The Amazon forests are likely to have a disproportionate effect on the cycling of global greenhouse gases. A typical Amazon forest is cycling carbon dioxide (CO_2) through metabolic respiration and sequestration CO_2 via photosynthesis. These metabolic processes occur throughout the year and only have a small dormancy period in areas that have seasonal drought. This period of metabolic inactivity is negligible relative to temperate forest's annual winter inactivity. In most cases, Amazon rain forest tree species are typically comprised of species with wood of a high specific gravity with many

species having specific gravities higher than that of water, *i.e.* the wood is so heavy that it sinks in water. These dense forests are also the largest expanse of tropical rain forests in the world. The combination of elevated metabolism, aseasonality and dense wood composition makes the Amazon Basin one of the largest atmospheric carbon banks for global carbon.

Unfortunately, this unique forest system also has the highest absolute rate of deforestation in the world (Skole & Tucker 1993, Laurance *et al.* 2001a).

Forests are typically cleared to provide open land for agriculture. In this process, felled forests are subsequently ignited releasing carbon into the atmosphere. Recent estimates equate that deforestation from tropical forests possibly represents 10% of the annual atmospheric carbon increase since the 1960's (Myers 1991, Fernside 1997 and Nepstad *et al.* 1999). Laurance *et al.* (2001a) have used data collected by the Brazilian space agency (INPE) responsible for tracking deforestation rates. They conclude that, despite a deforestation reduction occurring in the early nineties, deforestation rates have again increased to the previous levels occurring in the 1980's. This is a radical rate of clearing.

The extent and complexity of the Amazon has warranted the employment of data-capture technologies (GIS and remote sensing) to help characterize the ecosystem and land-use effects. Such methods were coupled with intensive ground data sampling in the present study. A brief overview of these techniques is discussed below.

DATA-CAPTURE METHODS

The advent of geographic information systems (GIS) and remote sensing have allowed for the development of accurate estimates of deforestation in the Amazon Basin (Skole & Tucker 1993, Houghton *et al.* 1999, Defries *et al.* 2002). GIS and remote sensing allow investigators to accurately quantify landuse changes that have occurred in these forested systems far more efficiently than traditional land based surveys. Also, the majority of the Amazon Basin is only accessible from a boat, airplane or pedestrian transport. Satellite based sensors are the best means to quantify the rates and location of deforestation. It is thus necessary to review the capacities and methodologies used by computer based mapping software programs.

GEOGRAPHIC INFORMATION SYSTEMS

Geographic Information Systems (GIS) are cartographic and spatial analysis software programs. Attributes of map elements can be associated with individual cartographic objects. Collectively, these objects can be sorted, queried and manipulated based on their associated attribute data in similar fashion as data can be manipulated on common spreadsheet programs. A set of spatially explicit location data with their associated attributes is called a 'data layer' where data-layers can be combined, selected from or geometrically manipulated in a spatially explicit fashion determined by the methodology used.

Data Layers can come in a variety of different geometric configurations. Point data is a simple location (a x, y coordinate location) with associated attributes. Line data is a set of vertices confined between two endpoints. Line-data also have attributes

assigned by the author, values for the left or right as well as the direction of the line.

Lines can be assembled to form networks to investigate processes such as flow, cost/benefit analysis and the shortest time to travel from location 'a' to location 'b.' Polygons are a set of lines that enclose an area. This area has associated attributes derived from the author, area and perimeter measures. Points, lines and polygons are called vector data and are typically used together.

Vector data can be manipulated spatially. One method allows the user to buffer selected vector data by a determined distance. In the event that the data is polygonal buffers can be generated from the outside edge outwards or from the outside edge inwards. Vector data can also be 'overlaid' with other vector data to form new vector data that combine the attributes of the original data layers into one data layer. By combining data layers, it possible to sort and query data in complex but spatially explicit fashion. For example, buffers could be drawn around a linear watercourse to represent areas that are critical to water quality. Not all of the areas surrounding the watercourse are in danger since some are protected conservation areas. By combining the river buffer with the conservation polygon data layer, the user can parse out areas that are not protected and of more immediate interest to water quality protection.

Another type of GIS data is called grid data. Grids are squares or rectangles of equal area in an array of x cells wide and y cells high, *i.e.* a grid cell array. A single grid data layer has only one numeric value associated with each cell. An integer value can represent anything and exact statistics can be generated concerning the frequency of values. Numeric values represent a value within a continuum of values and descriptive statistics can be generated but not to the precision of integer values. Similar to vector

data, grids can be combined from several data grids to form a new data grid based on the combination of data layers or mathematical operations between grids. Vector and grid data are interchangeable but considerable distortion of the data can occur due to the immutability of grid cell sizes.

REMOTE SENSING

Remote sensing is the field of study that involves using imagery to quantify or qualify environments in which the researcher does not have direct contact. A familiar 'remote sensing' project would delineate habitats using aerial photographs. Field locations are visited both to help calibrate (or train) interpretation as well as provide for verification after the delineations are completed. Presently there are numerous types of sensors that are either flown *via* aircraft or space based satellite. Techniques and methodology are as diverse as the numerous sensors presently deployed from both satellite and aerial platforms.

Aerial photography (imagery) has a long history of use and development within the United States. The majority of National Resource Conservation Service Soil maps are just one end product of the use of aerial image interpretation. Different soil types are identified by the types of vegetation that occur on them. Many times, the presence or absence of a conspicuous vegetation species will identify the soil type being delineated. Traditionally, aerial photography was used as the primary imagery for these 'classifications.' Aerial photography has progressed from its initial use of standard black and white formats to color to the use of color infrared films. Infrared films are able to differentiate different vegetation types more accurately because of the more active

inflorescence from species chlorophyll and secondary photoactive pigments in the infrared wavelength. Recently, aerial digital imagery is replacing aerial photography making data cheaper to acquire and process. Aerial digital imagery has the added feature of being able to incorporate color (red, green and blue) bands with infrared creating multi-band spectral data. When viewing the imagery the user can choose three spectral bands to be viewed as red, green and blue. Greater accuracy in image interpretation is possible in most regions.

Satellite imagery data has been available since the mid-1970's and is used for either large scale projects (10,000 ha+) or to classify areas that are beyond the reach of modern transportation. Nation Aeronautics and Space Administration's (NASA) Landsat 4 Multi- Spectral Sensor imagery (MSS, L4) pioneered the first space based satellite imagery available to non-defense based professions. The MSS sensor acquires data of an area once every two weeks and records the reflected light from earth. It has a minimum resolution of 90 x 90 m, meaning that the smallest area depicted is almost 1 hectare in size. This sensor records reflected light from earth in multiple wavelengths in visible and infrared bandwidths. Infrared is particularly useful in tropical regions since its spectral reflectance is an accurate indicator of habitat physiognomy. Habitats with low or flat stature reflect large signals in the infrared wavelength where heat (infrared light) is reflected back off the surface. The earliest studies quantifying the extent of deforestation occurring within the Brazilian Legal Amazon used this satellite based imagery to quantify Amazon rain forest distributions for 1978 (Skole & Tucker 1993). In 1984, the next generation Landsat satellite became available for use, Landsat 5 Thematic Mapper imagery (TM). TM imagery uses a primary resolution of 28.5 x 28.5 m resolution for

most spectral bands (band 6 of the very far infrared is 90 x 90 m). TM differs from MSS in that the infrared wavelength sensors are broken into near, middle, far and very far infrared wavelengths. The dissection of infrared wavelength accompanied by the smaller resolution allows for more accurate interpretation and classification. TM is also one of the most widely used and explored satellite imagery available. In 1999, Landsat 7 Enhanced Thematic Mapper (ETM+) imagery started to be acquired. L7 imagery data is very similar to TM imagery with the added benefit of more precise reflectance sensors and an added panchromatic band with a 15 x 15 m resolution. The added panchromatic band is very useful in urban landscapes but does not aid in the improvement of forest/non-forest classification within tropical environments.

Satellite imagery is classified from multiple bands of spectral information into single band classified data, *e.g.* land use types, via multiple methodologies that statistically group the spectral data. A single band of Landsat data (MSS, TM and ETM+) comes in a '8-bit format' meaning that the light reflectance for that particular wavelength is quantified on a scale from 0 to 255. Higher values represent increased reflectance. The main spectral bands used to classify TM and ETM+ imagery are the red, near infrared and middle infrared, but one can use as many bands or as few bands as they deem reasonable. The first type of classification to be reviewed is called unsupervised classification. The user specifies the number of different classes they would like to differentiate. The unsupervised classification program then analyses the frequency distributions of each band and between bands and classifies the data into the appropriate number of classes based on a division scheme that minimizes class misclassification. Unsupervised classifications are primarily used for data exploration and are not generally

used for peer reviewed publication. In supervised classification methodologies, the user displays an image using three spectral bands (but the classification can use more or less). Polygons are manually delineated around visible features that the user can positively identify. Multiple polygons are used for each land use class. The spectral information for each band being used is collected for cells that the user delineates. The frequency of the spectral values within all bands selected is then characterized via descriptive statistics for all of the cells within the class the user wished to identify (mean, range, variance, etc). When the user is satisfied with the number of classes s/he has identified, the program classifies each pixel (or cell) according to the values it contains. The user can specify whether they wish to classify just the pixels that fall directly within the statistical range of possibilities or choose to classify all pixels using a maximum likelihood or other classification methods.

Since remote sensing programs are also capable of incorporating spatially explicit location data, thus have similar capabilities as GIS grid functions. Remote sensing of satellite imagery is the primary means by which tropical rain forest deforestation is identified. If these spatially explicit functions are incorporated into the analyses, remote sensing can not only quantify the total amount of area deforested but also identify where the deforestation is occurring. At present, INPE derived statistics on deforestation are only identified to the state level and more precise and accurate methodologies are not used to identify locations (INPE 2001).

Before this research, there has been a significant amount of field and remote sensing analyses. Background information is described that provided the foundation and stimulated the specific focus of the analyses.

AMAZON FOREST STRUCTURE AND DYNAMICS

The Amazon watershed is comprised of forested and non-forested habitat types. The non-forested habitat types are mainly peripheral *cerrado* (a scrubby habitat to the southeast) and higher elevation savannas that occur in the Guyana highlands and just interior to the north Atlantic coast. Drier and seasonal forests exist in the southern Amazon and have a more open canopy relative to the moist forest that dominates the watershed. Closed canopy tropical moist forests occur with multiple canopy strata in most of the upland forested habitats. These closed canopy forests can be further divided into plateau forests, downhill forests, shorter canopy *campinarana* forests and bottomland forests as one descends towards the river (Ribeiro *et al.* 1999). All forests that are not inundated are called *terra firme* forests. Adjacent to major rivers *várzea* forests occur. These forests are inundated with up to several meters of water annually. As a result the forest understory is much more open and the tree species that inhabit these areas have hydrochorous or ichthyochorous fruits.

FOREST STRUCTURE

Amazon *terra firme* forests can be seen as having a complex multiple tiered vertical canopy structure (Terborgh 1992). In ideal situations, there are as many as six levels to this forest structure that is dictated by the spatial relationships between the upper canopy and the movement of the sun as it migrates between the tropics of Cancer and Capricorn (23° N & S latitude, Terborgh 1992). The largest trees in the forest are termed emergents. These large trees extend above the canopy by 10 to 15 m and are capable of withstanding the increased stresses of microclimate and wind shear. These large trees

also serve a habitat for climbing vines as well as epiphytic bryophytes. One species existing in the central Amazon, *Ceiba spp.*, has wood that can not be harvested without silicon carbide tipped saws since its wood is denser than the saw's metal blade. Studies have found that up to one third of a forest's biomass can be held in these super-emergent trees that occur at a frequency of less than one individual per hectare (Rankin de Merona *et al.* 1992).

The next layer is the upper canopy where the tops of tree form a carpet of photosynthetically active leaves. Tree architecture is to spread the canopy out over as large an area as possible in order to acquire valuable sun light without growing vertically above the canopy in order to avoid increased wind shear. The lower canopy stratum is often structurally 'messy' and open to interpretation. It has slow growing, low light adapted tree species present (Laurance *et al.* 2004). The upper middle canopy occupies a space that is about 5 – 10 m below the upper canopy. It is occupied by sinuous trees that are not occupying canopy space but are in the position to gain canopy space when it become available. These trees are at a metabolic balance where all excess resources have been allocated to vertical growth at the expense of storing the energy in a re-useable form, *e.g.* carbohydrate. Due to the large amount of biomass put into vertical growth a high percentage of the light energy captured is respired shortly. Due to this severe imbalance no excess energy reserves are available for severe weather conditions, parasites or injury.

Approximately 7 m below this layer, there is the first tier of the middle canopy. These trees are of a much smaller stature (10 – 15 m) and are waiting for more light resources to become available from a gap initiation in the upper middle canopy. These

trees are under considerable physiologic stress where many do not grow but simply acquire enough photosynthate required for respiration. Trees that exist in this suspended state for more than a century (Terborgh 1992). The next lower canopy layer occurs at below 3 m, where seedling stature vegetative growth is prevalent. Although these trees have a diminutive stature their age can be large if they have enough light energy to sustain metabolic needs. Interspersed in this layer small stature palms become more prevalent. Finally there is a ground layer, which may or may not occur, depending upon local edaphic and climatic conditions.

This complex vertical structure ensures that all available light energy is converted into biomass. This structure also allows for complex behaviors that can occur at different strata in the canopy. One example would be where there is a strong dry season of long length. Canopy occupants sometimes will shed their leaves in order to decrease their 'epiphyte' load that decreases the efficiency of photosynthetic surfaces in times of physiological stress. The dehiscence of these leaves allows more light energy to penetrate the middle canopies where previous physiologically stressed trees will not shed their leaves but capitalize on the increase in light energy available to them even though it may be limited due to unavailability of water (Terborgh 1991).

TREE FALL GAP DYNAMICS

Like other systems, the Amazon forests regenerate *via* single or multiple tree fall gap replacement in natural systems. In the typical scenario a tree large enough that it has attained canopy placement senesces and dies. When the tree falls to the ground it is likely to collide with other trees either having canopy placement or occupying the middle

and lower canopies. The resulting canopy space is likely to be a semi-linear dumbbell shape resulting from the space that falling tree occupied and the linear felling of trees in the path of the tree as it crashes to the ground (Figure 1.2). Depending on the architecture of surrounding trees relative to the newly formed canopy gap and the actual size of the gap, the process of gap closure ensues *via* two processes. Since canopy space is limited and valuable to individual trees, adjacent trees occupying the canopy are likely if possible to extend their crown into the adjacent free space. This allows the individual to garner more light resources with limited metabolic cost, *e.g.* the tree doesn't have to grow a supportive trunk in order to occupy the space. Occupying this adjacent open space has an architectural cost. If the tree occupies the space successfully and the space is far from the trunk, the tree is more likely to become unbalanced. When unbalanced the tree is susceptible to wind shear and has a higher possibility that it will fall to earth, forming a larger canopy gap. Another likely progression to gap closure is for a tree left occupying the middle canopy. This tree might have enough stored resources to be able to grow quickly into the free canopy space and out compete its neighbors.

Turnover rates, *i.e.* the time it takes to replace all trees within a forest plot, have been estimated to be between 82 and 204 years (Lovejoy *et al.* 1986, Rankin de Merona *et al.* 1992) at the Biological Dynamic of Forest Fragments Project in the central Amazon (BDFFP). Turnover can be viewed as the average length of time that an individual tree will occupy a place within the forest. Higher turnover rates are representative of more mature 'old growth' forest compositions. This measured turnover rate is likely to be an overestimate since it includes forest edge habitat (Laurance *et al.* 1997, Laurance *et al.*

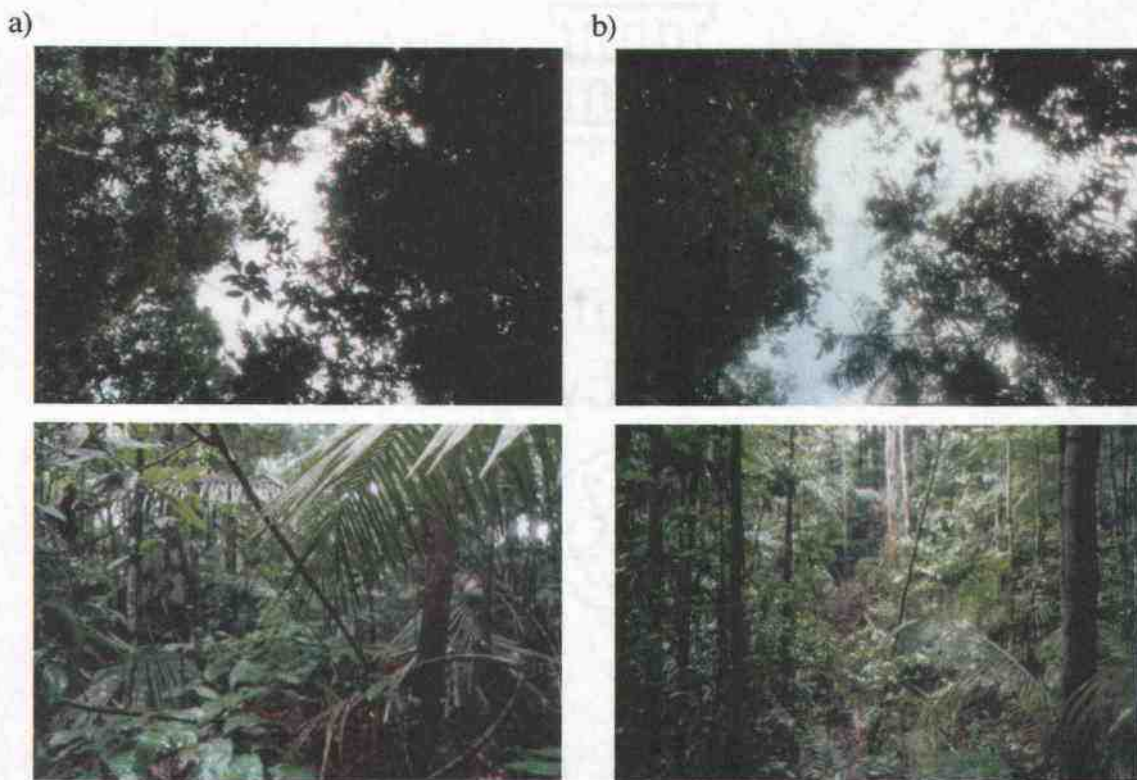


Figure 1.2 Canopy gaps and understory from tree falls within a central Amazon *terra firme* forest. Linear gap formations are created from one tree falling with subsequent trees within the falling trees trajectory being injured or killed. Both gaps (a and b) have linear forms. The understory of gap b clearly shows a linear zone of mortality on the ground.

2000). Phillips and Gentry (1994) studied tropical rain forest turnover rates and found that turnover rates were increasing worldwide, even in 'pristine' western Amazonian forests. They speculate that this increase is ultimately due to the effect of increased CO₂ concentrations upon tree growth and mortality rates.

Larger scale disturbances occur from thunderstorms and their associated 'blowdown' downdrafts (Nelson *et al.* 1994). These tree fall blowdowns are larger than the tree fall gap dynamics described above and occur in discrete events where trees are denuded and uprooted. This Brazilian Amazon study consisted of 137 TM scenes in which blowdowns were identified if they were larger than 30 ha. Nelson *et al.* were successful in locating three large blowdowns occurring near *Manaus* that were initially identified on TM imagery (28.5 m² resolution). Most of the blowdowns were in the smallest size class but the area of blowdowns occurs across all size classes (30 - 3,400 ha) with the area slightly skewed to smaller classes less than 1,000 ha. When smaller blowdowns (< 30 ha) were accounted for in two TM scenes, the frequency of blowdowns increased by 10 fold. From the data collected, Nelson *et al.* (1994) estimated that the turnover rate for these large scale disturbances to be 5,000 years for the whole Brazilian Amazon. Turnover rates are higher in areas that concentrated blowdown events on the 'interfluve' of the *Japurá* and *Negro* rivers occurring along a north- south axis. This also was spatially correlated with areas that had 25% of their days receiving more than 20 mm of rainfall, indicating thunderstorm 'downbursts' were the main cause of this disturbance.

ANTHROPOGENIC EFFECTS UPON FOREST DYNAMICS

Most of the deforestation occurring within the Amazon Basin is not due to natural disturbance but is from non-endemic anthropogenic sources. When mature forests are cleared, edge effects have a demonstrable effect on the pattern and processes of forest within 300 m of the forest edge (Lovejoy *et al.* 1986, Ferreira & Laurance 1997, Laurance *et al.* 1997, Laurance *et al.* 1998a, Laurance *et al.* 1998b, Laurance *et al.* 2000, Laurance *et al.* 2002b). In forest fragments isolated from mature forests in the early 1980's biomass was shown to decrease by as much as 36% within 100 m of the forest edge (Laurance *et al.* 1997). The most probable cause of this increased mortality is due to increased windfalls, changes in microclimate that promote drought stress and increases in parasitic liana growth. Large trees (> 60cm diameter breast height, DBH) are disproportionately killed within 300 m of the forest edge (Laurance *et al.* 2000), although all size classes have increased mortality. Large trees represent only 1.8% of the number of trees with DBH > 10 cm but are nearly a quarter of the total biomass (23.4%). Tree recruitment to edge habitats favored ruderal pioneer species such as *Cecropia* and *Vismia spp.* and mature species recruitment was inhibited (Laurance *et al.* 1998b). Conversely, regrowth forests within 100 m of mature forest edges have a higher species richness (Mesquita *et al.* 2001).

DEFORESTATION RATES WITHIN THE BRAZILIAN LEGAL AMAZON

One of the most referenced study of deforestation within the Brazil Amazon was conducted by Skole & Tucker (1993). This study quantified the deforestation rates by detecting the differences between 1978 (MSS) and 1988 (TM) satellite imagery. The study delineated deforestation within the imagery in a GIS vector format for each year. Land change analysis had shown that deforestation occurred at an average rate of 15,000 km² per annum. An additional 38,000 km² per annum was compromised by increases in fragmentation, decreases in biological diversity and habitat degradation from increases in non-endemic human intrusion. The estimate was lower than those proposed previously but only represents actual 'clear-cut' forestry and forested areas within 1 km of forest edge. By 1988, 6% of the forested Brazilian Legal Amazon was deforested and 15% of the forested areas were compromised by anthropogenic intrusion. The states of *Para*, *Mato Grosso* and *Rondônia* had the largest increases in deforestation, habitat isolation and areas affected by edge effect. Nepstad *et al.* (1999) studied the deforestation within southern Brazilian Amazonia and found that up to 60% of areas identified as forest in TM imagery had incurred selective cut forestry harvest and understory fire events. This finding, if consistent throughout the entire Brazilian Legal Amazon, would mean that Skole and Tucker's (1993) estimates of fragmented and degraded habitat were under evaluated. Since the Nepstad *et al.* (1999) study occurred in an area of particularly high deforestation, *i.e.* 'the Arc of Deforestation,' these findings do not extend past eastern and southern Brazilian Amazonia, making the need for conservation more dire within these areas of high deforestation activity.

CARBON DYNAMICS OF THE BRAZILIAN LEGAL AMAZON

After performing multi-temporal analyses of tropical deforestation in the Brazilian Amazon, Houghton *et al.* (1999) developed a 'bookkeeping' model estimating the potential inter-annual variability in atmospheric carbon contribution (Houghton 1999b). A model performs a spatially explicit comparison of two years, 1986 and 1992. In intervening years, statistics gathered by INPE are used (these statistics are obtained by manually delineating of deforestation occurring within each of the states of the Brazilian Amazon). The spatially explicit data from 1986 and 1992 were used to 'calibrate' the INPE statistics that routinely overestimated deforestation occurring within the Amazon (by 25% - 50%, Skole and Tucker 1993, Houghton *et al.* 1999, Defries *et al.* 2002).

Biomass of mature forests was interpolated from RADAMBRASIL (Brazilian forest inventories) and then an area weighted biomass was created for each Brazilian state. This estimated value was also increased by 60% to account for palms, vine, deadwood biomass and belowground biomass. Biomass was also interpolated from 56 plots where biomass data was collected indirectly and directly. Deforested biomass was committed to ignition, decay, removal for products and conversion to 'elemental carbon' at the following ratios 0.2, 0.7, 0.08 and 0.02. The decay and removal for products biomass was estimated at 0.1 yr^{-1} . Elemental carbon decayed at a rate of 0.001 yr^{-1} . Deforested areas are either committed to agriculture or progress into secondary forest once they are abandoned. Regrowth forests sequester atmospheric carbon at a rate of 2.8% per year of their mature forest biomass for the first 25 years then subsequent sequestration is lowered to 0.6% per year until year 75. The Bookkeeping model tracks the amount of carbon lost to ignition, decay, removal for products and elemental carbonization for each year based

on yearly estimates from INPE (except for 1993). Biomass contributed to ignition is immediately released whereas decay, removal for products and elemental carbon decay over time. Coinciding these biomass contributions to the atmosphere, the Bookkeeping model tracks the amount of carbon being sequestered on fallow agricultural lands. The difference between the annual emissions and annual sequestration is the net annual flux of carbon. This model found that $0.2 \text{ Pg Carbon yr}^{-1}$ was the average contribution for the 1989-1998 study area (1 Pg is 10^{15} g). They found that the inter-annual variability varied around the mean by 0.2 Pg C yr^{-1} with an additional 4 - 7% needed to be added from logging and another 0.2 Pg C yr^{-1} from drought year fires. The researchers conclude that there is a great need for more spatially explicit multi-temporal models to get clearer estimates of regrowth forest tenancy and its effect on carbon budgets. The need for more spatially explicit multi-temporal models was again confirmed in a similar bookkeeping model of all of the tropical rain forests (DeFries *et al.* 2002). Chapter three of this manuscript creates such a model and reports its results for carbon budgets in the central Brazilian Amazon.

Since these extant studies, the future of the Amazon has entered an accelerated phase of anthropogenic development. The pressures of continued deforestation, land conversion, attendant environmental degradation and road incursions are creating intense and probably irreversible losses and change in the Amazon. Some of these trends are discussed below.

THE FUTURE OF THE BRAZILIAN LEGAL AMAZON

Laurance *et al.* (2001b) developed a spatially explicit model that sought to account for major new development projects brought into the Brazilian Legal Amazon under a federal program called *Avança Brasil* (recently renamed *Pluri-annual 2004-2007*). Their analysis accounts for new road upgrades, roads, railroads, utility corridors, hydroelectric projects, river canalization, mining, timber concessions, forest fires and riverside deforestation. This analysis also included the protection provided by high and moderate protection conservation reserves as well as recognized delineated indigenous lands. Using previous deforestation that had occurred once roads were established, a deforestation rate was quantified for areas within distances 0 - 10, 11 - 25, 26 - 50 km from the road edges for all roads (paved and unpaved) that existed within the Brazilian Legal Amazon (IGBE 1995). Paved roads (improved roads) had a higher rate of deforestation relative to unpaved roads. Using the parameters derived from the road deforestation, two spatially explicit models were developed to predicting deforestation into the year 2020. These models predicting the spatial distribution of pristine forests as well as lightly, moderately and heavily degraded forest habitats occurring within the Brazilian Legal Amazon.

The optimistic scenario predicts that there will be continued deforestation in the southern and eastern portions of the Brazilian Amazon, and considerable large-scale fragmentation of forests in the central and southern parts of the basin. The Brazilian Amazon will be nearly bisected by heavily to moderately degraded areas along a north-south axis running from *Rondônia* to *Manaus* and northward to Venezuela. Pristine and lightly degraded forests will be fragmented into several blocks, with the largest tract

surviving in the western Brazilian Amazon. According to this scenario, pristine forests will comprise just 27.6% of the region, with lightly degraded forests comprising another 27.5%. Almost 28% of the region will be deforested or heavily degraded.

The non-optimistic scenario projects an even more dramatic loss of forests along the southern and eastern areas of the basin. Large-scale fragmentation will also be more extensive, with much forest in the central, northern, and southeastern areas persisting only in isolated tracts. There will be very few areas of pristine forest aside from those in the western quarter of the region. This scenario predicts that pristine forests will comprise just 4.7% of the region, with lightly degraded forests comprising another 24.2%. Nearly 42% of the region will be deforested or heavily degraded. In either extreme, the nature of the Amazon will be severely changed.

DISSERTATION TOPICS

The major theme of this dissertation is the analysis and modeling of spatio-temporal forest dynamics in the Amazon across scales using data-capture technologies. GIS and remote sensing coupled with intensive ground sampling was used to investigate spatial relationships in forest dynamics occurring within the Brazilian Legal Amazon. The topics covered follow a spatial hierarchy from smallest to greatest in the spatial scale where they occur. The first investigation seeks to quantify the effects of landscape composition on regrowth forest tree species recruitment patterns. The process of seedling establishment depends upon seed transport and microclimate at the establishment site both of which are influenced by spatial configuration. This research investigated the importance of conserving large forest tracts and the consideration of spatial design in the preservation of biodiversity.

The second line of enquiry in the dissertation studied forest inter-annual variability and carbon budgets of the very dynamic central Amazon landscape. Inter-annual variability refers to the anthropogenic uses of mature forest and regrowth forests. Mature forest deforestation is well studied within the Amazon (Skole & Tucker 1993, Houghton *et al.* 1999, Laurance 2001c, Defries *et al.* 2002 and INPE semi-annual reports concerning deforestation). The dynamics of secondary forests within the Amazon basin are less well understood (Houghton *et al.* 1999, Defries *et al.* 2002). Through the development of a multi-temporal Landsat analysis (MSS, TM and ETM+) a chronosequence of anthropogenically induced deforestation and regrowth forest land tenure is quantified in terms of carbon emissions, carbon sequestration and land tenure cycles thus

yield a more precise and accurate understanding of these previously ‘guestimated’ processes (Houghton *et al.* 1999).

The final and third topic of investigation within the dissertation is the quantification of the amount of deforestation to be expected from each of the development mechanisms contained within Laurance *et al.*’s (2001b) predictive land use model for the Brazilian Legal Amazon. Using spatial regression models, a methodology is developed to quantify the maximum and minimum deforestation, habitation degradation and fragmentation caused by each of the multiple development types contained within *Avança Brasil*. The methodology also quantifies the amount of protection provided by high and medium protected reserves systems and indigenous lands in light of these development scenarios. Previous studies have concentrated on the protection management level of protected areas within the Brazilian Legal Amazon (Peres and Terborgh 1995) but not upon the actual amount of forest saved from development.

An overview of the approach taken by each of the three studies is presented below.

THE EFFECTS OF LANDSCAPE COMPOSITION ON SECONDARY FOREST TREE SPECIES RECRUITMENT

Regrowth forests (secondary forests) have been of interest in tropical rain forest systems since the middle 1980’s (Uhl *et al.* 1988, Saldriga *et al.* 1988). Most of these studies have centered upon the rate at which these species sequester atmospheric carbon in relation to land use history (Uhl *et al.* 1988, Saldriga *et al.* 1988, Mesquita 1995,

Goody *et al.* 1996, Steininger 1996), their rates of sequestration across tropical gradients (Steininger 2001) and how species composition affects biomass production (Steininger 1996, 2001, Mesquita *et al.* 2001). The factors influencing secondary forest species richness are the age of the secondary forest plot (Saldriga *et al.* 1988) and the plot's proximity to mature forest stands (Mesquita *et al.* 2001). Studies have been conducted that use satellite remotely sensed data to quantify and estimate regrowth forest ages (Goody *et al.* 1996) and to qualify the existing immediate landscape (Steininger 2001). Studies seeking to interpret the interactions between nearest forest and time since regrowth forest initiation have not been conducted. Chapter Two quantifies the interaction of secondary forest age, proximity to mature forests and the condition of the surrounding forest. It also investigates how the type of how mature forest condition is important to influencing the richness of these secondary forests and estimates the time until these regrowth forests recover a full tree species component that is found in adjacent mature forests.

INTER-ANNUAL VARIABILITY AND ITS INFLUENCE ON THE MEASUREMENT OF TOTAL ABOVE-GROUND BIOMASS

Recent studies of the Amazon basin's contribution to global greenhouse gas emissions are calculated by estimating the biomass lost from the ignition of previous mature forests (Skole & Tucker 1993, Houghton *et al.* 1999, Defries *et al.* 2002). Carbon released into the atmosphere from mature forest ignition has been estimated to be approximately 10% of the global total annual increase since the rapid development within the Brazilian Amazon started in the middle 1960's (Fernside 1997, Laurance *et al.* 1998,

Nepstad *et al.* 1999, Houghton *et al.* 1999, Defries *et al.* 2002). Small-scale regrowth forest dynamics have also been studied on the scale of individual family plots (Browder & Godfrey 1997, Pfaff 1999). These smaller scale studies have been centered on the economic resource utilization and not necessarily equated to carbon dynamics. Studies concerning regrowth forest carbon budgets have equated emissions from a varied number of accounting methodologies but none have measured the contribution of land use tenure of secondary forests in a spatially and temporally explicit fashion (but see Houghton *et al.* 1999 for the most accurate to date). Both Houghton *et al.* (1999) and Defries *et al.* (2002) mention the need of spatially explicit models that incorporate spatially explicit data that account for inter-annual variability in a more precise and accurate method through time. In Chapter Three a methodology is developed which tracks the annual occupancy of individual pixel cells (30 x 30 m) in a dynamic anthropogenically influenced landscape. The model tracks deforestation rates and land use tenure subsequent to deforestation as well as all pixels that were altered previous to 1976 (the initiation of the model). This model uses spatially explicit Landsat (MSS, TM and ETM+) remotely sensed satellite imagery for the years 1976, 1985 - 1997 and 1999. This model quantifies areas that are sources of atmospheric carbon and areas that are sinks (sequestering carbon from the atmosphere). Quantifying sources and sinks of atmospheric carbon within dynamic landscapes testifies against the previous static estimations of inter-annual variability based on broad generalizations that do not reflect the changing anthropogenic land tenure across the Amazon Basin through time. Such attention to land tenure dynamics and heterogeneity question the validity of simple static

models and suggest that inference based on large-scale modeling is highly uncertain and must be evaluated cautiously.

CONTRIBUTIONS OF DEVELOPMENT PROJECTS TO FUTURE DEFORESTATION WITHIN THE BRAZILIAN LEGAL AMAZON

Laurance *et al.*'s (2001b) prediction of future deforestation within the Brazilian Legal Amazon includes future deforestation occurring from planned development projects that includes road improvements, new roads, utility corridors, hydroelectric projects, new railroads, forest fires, natural resource extraction and riverside deforestation. This model also includes areas that will prevent deforestation, high protection reserves, and areas that will decrease the amplitude of deforestation such as moderate protection reserves and recognized indigenous lands. One of the major criticisms of this interactive continental scale model is that "it is impossible to decipher the relative amounts each project type contributes to deforestation" (Nepstad *et al.* 2001). Though not "impossible," disentangling complex models that have inter-reactive synergistic elements is as complex as the spatial model's creation. Chapter Five disentangles the published optimistic model published by Laurance *et al.* (2001b) and quantifies the maximum and minimum contributions of each development type as well as the amount of area that is saved by the present distribution of nature reserves. It also provides insight to how re-engineering the locations of infrastructural developments and the placement of sustainability reserves can constrain the deleterious effects of planned development within the Brazilian Legal Amazon.

Landscape Characteristics Influencing Secondary Forest Tree Species Richness In Central Amazonia

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ABSTRACT

Amazon tropical rain forests are being harvested at an increasing rate. Most of the areas that are harvested are for 'swidden' agricultural systems that are active for a short period of time followed by longer period of abandonment. It is not clearly understood how secondary forests occurring on previously abandoned land will succeed into mature forests. This study documents naturally occurring 'blowdown' events that can be as large as 70 ha in the central Amazon. The blowdowns are roughly the same size as the majority of 'swidden' agricultural tenure. Using abandoned agriculture plots that have succeeded to regrowth forests, we investigate the influence of secondary forest age, nearest mature forest, nearest non-fragmented mature forests and these forest's spatial attributes for their effect on tree species richness in adjacent secondary forests. $\log_{(10)}$ nearest forest size, $\log_{(10)}$ distance to non-fragmented forest, agriculture plot size, and the age of the regrowth forest are shown to have significant associations. Step-wise multiple linear regression analyses are performed in order to quantify the relative influence of each of the variables. The $\log_{(10)}$ distance to non-fragmented forest is highly significant with all other variables being excluded from stepwise forward techniques. Stepwise backward multiple linear regression found that regrowth forest age and distance to non-fragmented forest are the most significant associations with regrowth forest tree species richness. Previous land tenure that burns mature forest or secondary forest did not significantly affect regrowth tree species richness when corrected for the effects of age of regrowth forest, distance to non-fragmented forest, or $\log_{(10)}$ distance to non-fragmented forests. Regrowth tree species richness is elevated when these forests reach seven years of age and are within 600 m of non-fragmented forests. Using a different

spatial analysis technique, the effect of non-fragmented forests on regrowth tree species richness is shown to be stronger than that of all forests (fragmented and non-fragmented forest combined). These results suggest that the importance of forests that contain full biologic integrity are necessary for the persistence of these highly species diverse forest systems. Non-fragmented forests have higher tree species richness that produces a more diverse seed source and contains a higher capacity to disperse seeds *via* mammalian and ground avian dispersal vectors. The age of regrowth forest is responsible for providing the architectural structure necessary to dampen microclimatic variables physiologically necessary for germination and seedling establishment.

INTRODUCTION

Amazon tropical rain forest systems are known for the biological species diversity they contain (Erwin 1983), the bio-complexity that sustains these forests (Terborgh 1988, Terborgh *et al.* 2002) and their contribution to global atmospheric processes (Houghton *et al.* 1999, Defries *et al.* 2002). While the exact contribution of these forests is not known, it has been estimated that they contain as many as 50×10^6 species (Erwin 1983) representing approximately one third of the species on earth (The Nature Conservancy *pers comm*). Currently the Brazilian Amazon forest systems are being harvested at an increasing rate (Laurance *et al.* 2001, INPE 2002, Nepstad *et al.* 2002). *Instituto Nacional de Pesquisa Espaciais* (INPE 2002, the Brazilian federal space research program responsible for monitoring deforestation within the Brazilian Amazon) studies state that most of this increase in deforestation results from small scale, family based 'swidden' agricultural expansion.

Clear-cut forestry, once prevalent in the 1970's and 80's, has given way to methods of selective cut forestry (Nepstad *et al.* 2002). Clear-cut forestry practices were used to establish large *fazendas* for beef cattle production (*fazenda* is Portuguese for cattle ranch). Many remote *fazendas* were not economically viable without government economic incentives. Without such financial incentives, many *fazendas* were abandoned and have subsequently had their pastures succeed into regrowth forests. Regrowth forests of the central Amazon have two successional pathways that are dependent on their previous land management (Mesquita 1995, Steininger 1996, Honzák *et al.* 1996, Williamson *et al.* 1998, Mesquita *et al.* 1999, Steininger 2001, Lucas *et al.* 2002). Areas that have had a tenure system that uses fire to suppress tree growth succeed into *Vismia*

spp. dominated forests (mainly *V. guianensis*, *V. japorensis* and *V. cayennensis*). Areas that have not utilized fire succeed into *Cecropia spp.* dominated forest (mainly *C. sciadophylla* and *C. purpurascens*) and occur less frequently (Steininger 1996, Lucas *et al.* 2002). Of these two secondary forest types, *Cecropia* dominated forests have a species composition that is similar to that which occurs from larger natural disturbances such as windfalls (Nelson *et al.* 1994, Mesquita 1995).

Uhl *et al.* (1988) studied the succession of secondary forests in relation to previous land use history. The main interest of these authors was biomass sequestered by secondary forests. The amplitude of previous land tenure influenced secondary forest biomass sequestration rates. Lands that were only lightly modified had the highest sequestration rates. Regrowth forests that were repeatedly burned, “moderately modified”, had a decreased sequestration rate for the first 10 years post-abandonment. Forests that occurred after severe land alteration, where the topsoil was mechanically removed, had a biomass sequestration rate that was severely limited and retarded 15 years after abandonment and beyond.

Saldarriaga *et al.* (1988) studied regrowth forest sequestration rates and the rate at which species richness increases with time in *tierra firme* forests of Guyana Shield origins. This study investigated regrowth forest plots that were of ages from 3 to 80 years. They found that agricultural plots are typically cultivated two to four years and then abandoned, followed by the establishment of regrowth forests that are dominated by pioneer species (*Vismia spp.* and *Cecropia spp.*). These pioneer species dominate up to 30 years when they senesce and other fast growing species replace them. These fast growing and persistent species dominate for an additional 50 years. As the fast growing

species senesce and canopy space opens, new species will occupy these vacancies but this is dependent on the size of the canopy gap. In areas of large canopy openings, *Cecropia spp.* can be expected to establish. In smaller gap openings, shade-tolerant species typical of mature forests will occupy the canopy gap. They found that more species accumulate as time progresses but no secondary forest stands had attained the diversity of the mature forest plots surveyed. Biomass was sequestered asymptotically, with older regrowth forests decreasing their rate of sequestration at about 30 years. Total aboveground living biomass was shown to equal that of mature forest at 189 years due to the poor soil nutrient conditions.

Hubbell *et al.* (1999) studied the effects of tree fall gap size (25 - 1,150 m²) to see if there was an influence in subsequent tree species recruitment richness in tropical rain forest occurring at Barro Colorado Island in Panama (BCI). Using a 50 ha tree plot in which individual trees greater than 1 cm stem diameter were surveyed every 5 years, they found that tree fall gap size did affect tree species composition but did not increase tree species richness when the data were corrected for the number of stems. The percent of stems that were pioneer species increased with larger canopy gaps. Conversely, shade-tolerant species decreased in the percentage of stems as canopy gap area increased. Hubbell (1999) states that one reason for the reduction of species richness in larger tree fall gaps is due to the mortality of shade-tolerant species. They conclude that tree fall gaps do promote tree species diversity regardless of canopy gap size but mainly by increasing the number of seedling establishment sites (Hubbell *et al.* 1999). The main factor allowing for high tree species diversity to persist through time is recruitment limitation. Recruitment limitation is the failure of a species to establish in all available

sites. If species are unable to recruit to all available locations, co-existence of many species is possible. It should be noted that tree fall gaps of this size (25 – 1,150 m²) are relatively small in comparison with large-scale blowdowns that occur within the central Amazon basin and have been documented to be as large as 3,400 ha (Nelson *et al.* 1994).

In 1997, an undocumented tree blow down occurred in proximity to the *Dimona Favenda* of the Biological Dynamics of Fragmented Forests Project (BDFFP) measuring over a continuous 70 ha (Figure 2.1, Bergen unpublished data). Blowdowns of this size are rare in the Manaus region but are of a scale comparable to medium sized ‘swidden’ agriculture prevalent in the central Amazon (Bergen and Steininger, in Chapter Two). Blowdowns of this scale are the largest natural disturbance within the central Amazon and serve hypothetically as a ceiling at which these forests are capable of recovering tree species diversity and biomass through natural processes.

Mesquita *et al.* (2001) conducted a study quantifying regrowth forest biomass within BDFFP 80 km north of Manaus. This study recorded tree species occurring along 100 m transects in regrowth forests aging from 4 to 18 years of age. They found that transects that occurred within 100 m of mature forest had an elevated species richness and that the increased tree species richness was most likely due to ‘seed rain’ occurring from passively and actively dispersed seeds from adjacent mature forest trees.

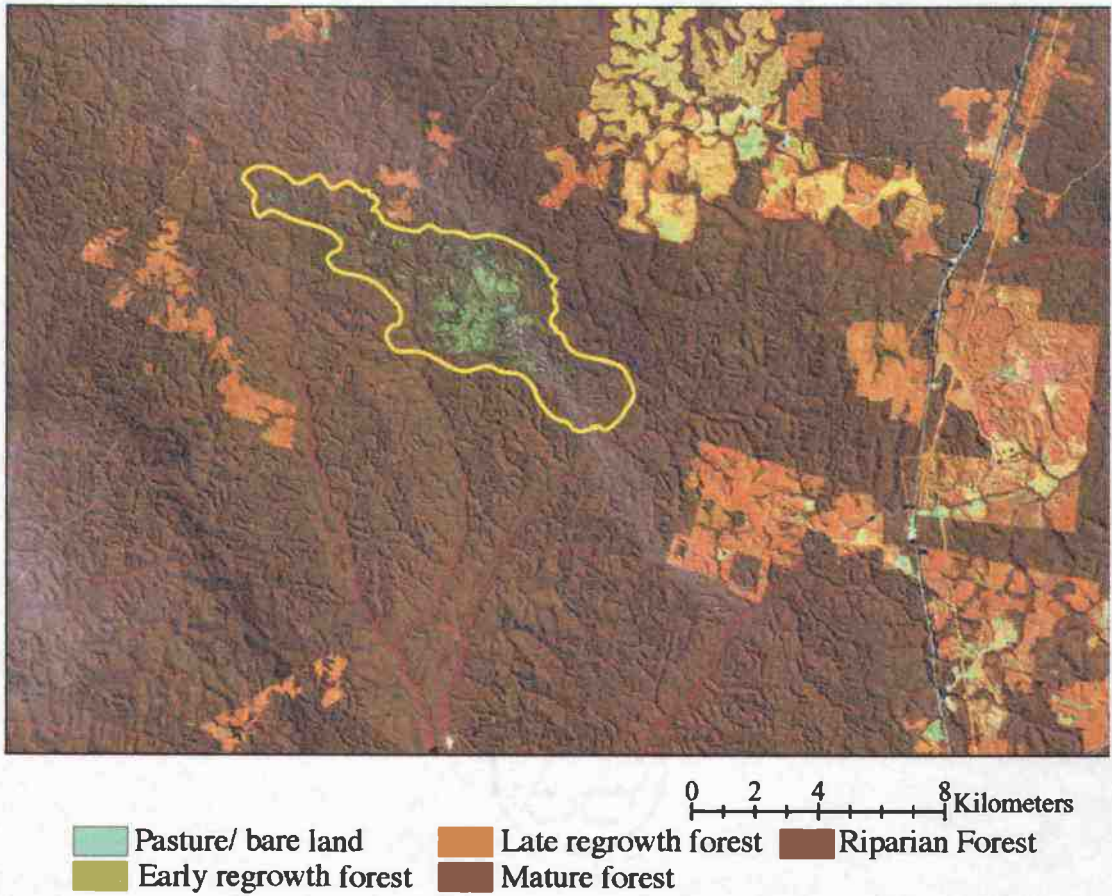


Figure 2.1 A 1997 Thematic Mapper satellite image of a large-scale blowdown measuring greater than 70 ha (area within thick yellow line). This image is a false color composite of bands for near infrared, middle infrared and red (4, 5 and 3) representing red, green and blue.

The examination of regrowth forest dynamics is fundamental to the development of sustainable forestry practices within the Amazon basin. This study investigates species richness in regrowth forests to qualify the large-scale disturbance processes and quantify how landscape characteristics, land tenure and disturbance area influence these ecological processes. One tenet of sustainability is that all species that exist within a forested landscape need to persist in the forest management's shifting mosaic of forest uses. By studying the factors that influence tree species recruitment, secondary forest composition, and temporal community richness patterns, foresters are much more able to devise management practices that are truly sustainable. By examining these factors, land managers will also be able to create forestry and silvicultural practices that have the potential to ameliorate previous unsustainable practices. Studies investigating tree species richness and turnover rates could determine how frequently a forest can be harvested while still sustaining high species diversity, bio-complexity and biological integrity. Without such sustainable forestry practices in place, we stand to lose a large proportion of species that inhabit these 'hyper-diverse' forest systems (Laurance 2001a).

MATERIALS AND METHODS

STUDY AREA

In 1993 and 1995, 29 100 m secondary forest plots were identified north of the city of Manaus (2°30'S, 60°00'W) in and about the Biological Dynamics of Fragmented Forests Project (BDFFP), Ducke forest reserve, and adjacent to paved road AM 10 to the east (Figure 2.2). Mature forests in this region consist of dense tropical forest with a canopy of 30 – 37 m with emergents reaching 55 m. These forests are very diverse and recent studies have calculated that as many as 280 species ha⁻¹ are typical with a high of 301 species ha⁻¹ recorded (Oliveira & Mori, 1999). Rainfall averages 2,105 mm annually within a range of 1,900 – 2,500 mm. There is a dry season from June to November during which 120 mm of rain occurs.

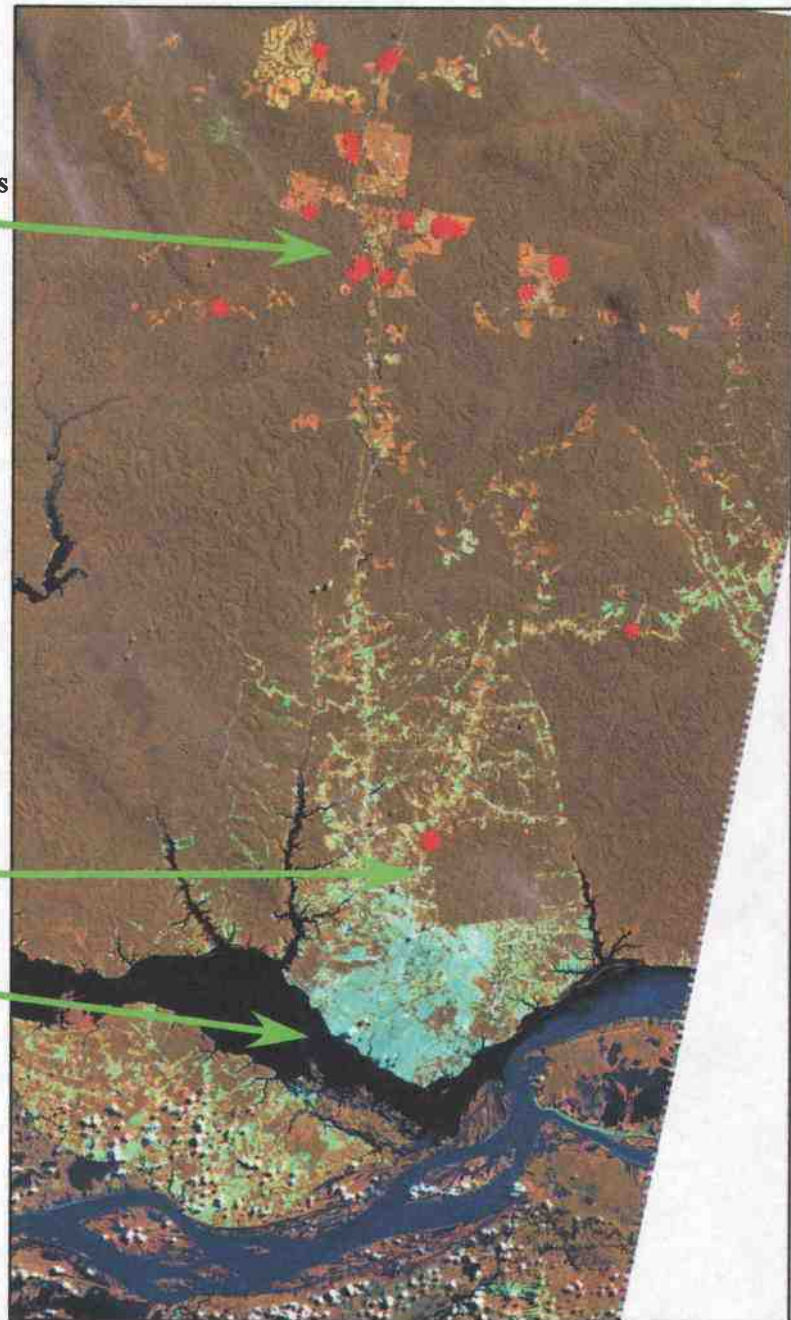
TRANSECT SITE SELECTION AND TREE SPECIES IDENTIFICATION

We located all regrowth forest tree transects *via* geographic positioning systems (GPS) to an accuracy of ± 30 m. We used a transverse mercator coordinate system to perform all subsequent data analyses. Transects were selected during the 1993 and 1995 field seasons. If a transect was re-sampled in 1995, the data from the 1995 field season was used. The direction of the transect was randomly chosen by selecting a number between 0 and 35. The selected number was multiplied by ten to yield the compass direction (zero to 350 degrees). If the compass direction resulted in the transect intersecting mature forest a new number was drawn. Transects were not located within 200 m of another transect. Transects were located where permission from the present

Biological Dynamics
of Fragmented Forests
Project

Reservo Duque

Manaus



Black water river
White water river
Cloud

Pasture/ bare land
Early regrowth forest

Late regrowth forest
Mature forest

Figure 2.2 The location of 29 100 m transect plots (red asterisks) occurring in secondary forests north of Manaus (2°30"S, 60°00"W) overlaid on a 1997 TM satellite image (4,5,3 = RGB).

tenant could be obtained and were within one-half days travel from metropolitan Manaus or BDFFP field camps. All trees within 1 m of the transect tape were measured, stem diameter, height and tree species were recorded. When an uncommon species was encountered, species were identified as best as possible with the help of curators of *Instituto Nacional de Pesquisas de Amazônia's* herbarium in Manaus (INPA). Using the morphospecies concept, 177 morphospecies were identified with 89 morphospecies not being identified to species, seven were not identified to genus. Age estimates were based on multi-temporal Thematic Mapper satellite imagery (TM) techniques (Lucas *et al.* 1993) or *via* interview with local residents. Areas within 6,400 m of transects were classified into physiognomic land use classes *via* visual interpretation of 1995 TM satellite imagery. These classes were water, pavement, pasture, early regrowth forest, late regrowth forest, and mature *terra firme* forest in a vector format. The resulting polygons were measured for their area and recorded in hectares.

VARIABLES AND STATISTICAL ANALYSIS

The following landscape characteristics relative to each plot location were measured, regrowth forest size (ha), continuous agricultural plot size (ha), nearest mature forest size (ha), nearest mature forest distance (m), distance to non-fragmented mature forest (m) and the age of the regrowth forest in which the plot was located (years). Regrowth forest size was measured by the area of the continuous even aged secondary forest patch in which the transect was located. Continuous agricultural plot size was determined by measuring the secondary forest patch of the transect, plus all adjacent secondary forests and agricultural lands until either mature forests or roads were

encountered. One agricultural establishment may have many different sections that are rotated in and out of production. Continuous agriculture plot size was estimated by the size of the total proximal land in recent production. Non-fragmented forests were defined as forests that were larger than 1,000 ha. Distance metrics were measured by the smallest distance between the 100 m transect and the mature forest type being investigated visually with GIS software.

All statistical analysis was performed using Statistica 6.0 © on a Windows XP operating system using a Pentium IV™ processor. All variables were tested for their effect upon the dependent variable (tree species richness) *via* single linear regression analysis to test the null hypothesis that the slope of the association between the independent and dependent variable is zero. All regression's residual *versus* predicted plots were visually analyzed for prevalent horn shapes and abnormal residual distributions. If a horn shape was apparent, a $\log_{(10)}$ transformation function was performed and inspected to meet the requirement of homogeneity of the residuals (Ramsey and Schafer 1997, Kéry and Hatfield 2003).

We entered all of the variables analyzed in the single linear regressions into multiple linear regression analyses to evaluate their strengths relative to one another and find a parsimonious regression model to explain the associations with regrowth forest tree species richness. We performed two multiple linear regression analyses, stepwise forward and stepwise backwards. A p value of 0.5 was used to enter or remove variables being evaluated. Since the stepwise forward analysis was initially performed by quantifying individual variables identical to single linear regressions, $\log_{(10)}$ transformations were used for the nearest forest size and distance to non-fragmented

forest variables. In stepwise backwards procedures, the initial evaluation of the variables occurs when all variables were included in one regression producing one set of residuals. These residuals were normally distributed and no transformations were required (Kéry and Hatfield, 2003).

The final multiple linear regression models of the stepwise forward and backward multiple linear regressions were then fitted with an indicator variable for previous tenure that included burning to test the null hypothesis that the slope of the association between land tenure and tree species richness is zero after accounting for the associations found significant by stepwise forward and stepwise backward multiple linear regression.

Age of regrowth forest and distance to non-fragmented forest were then graphically analyzed by a three dimensional plot and a two dimensional plots where the z axis is regrowth tree species richness or the z axis is expressed as isoclines of species richness. A quintic interpolation procedure was used to 'smooth' the surface between data points. A 95% confidence ellipse was generated by a non-parametric kernel density estimator based on the location of data points. This graphical procedure was used to interpret the extent of previous significant associations of the independent variables with regrowth tree species richness.

To increase the validity of non-fragmented forests association with regrowth tree species richness, a surrogate examination was constructed. A 400 m buffer was delineated around each of the 29 transects. The amount of fragmented, non-fragmented and total forest was quantified within the 400 m buffer. Since the 400 m buffer limits the measurement of the area of forest, the percent of the 400 m buffer within each of the forest groups was recorded. Because of abnormally distributed residuals in the analysis

of regrowth tree species richness and percent of fragmented forest, six cases were dropped from the analysis (two because they had zeros across all forest types and four because of their successive leverages). The independent variables of percent total forest within 400 m, total fragmented forest within 400 m, and total non-fragmented forest were tested via single linear regression against the dependent variable of tree species richness without the six cases mentioned to test the null hypothesis that the slope of the association between the independent variables and the dependent variables is zero. All independent variables were visually inspected for normal distribution of the residuals.

RESULTS

LANDSCAPE AND TRANSECT CHARACTERISTICS

Using simple linear regressions of species richness with each of the landscape variables, we found that many relationships were significant or highly significant. Nearest forest size is positively associated with tree species richness and is statistically significant ($p = 0.018$, $R^2_{adj} = 0.160$, Table 2.1). For each $\log_{(10)}$ integer an average increase of four more species (± 8.28 , 1 standard deviation) were found beyond the initial 32. Nearest forest distance did not show any significant relationship with the explanatory variable ($p = 0.462$, $R^2_{adj} = 0.016$, Table 2.1, Figure 2.3). The $\log_{(10)}$ distance to non-fragmented forest is shown to have a highly significant association with species richness ($p < 0.0001$, $R^2_{adj} = 0.467$, Table 2.1, Figure 2.3). If the model fits, for every $\log_{(10)}$ integer increase in distance from non-fragmented forest, 8.43 (± 8.83) species are lost from an initial 89. Regrowth forest size was found to influence tree species richness negatively but not significantly ($p = 0.076$, $R^2_{adj} = 0.079$, Table 2.1, Figure 2.3). Continuous agriculture plot size displayed a significant negative association with tree species richness ($p = 0.011$, $R^2_{adj} = 0.184$, Table 2.1, Figure 2.3). In the model, for every 1,000 ha increase in continuous agriculture plot size, 7.2 (± 14) species are lost from regrowth forest tree species richness. The age of regrowth forest is highly significant in its association with tree species richness ($p = 0.0001$, $R^2_{adj} = 0.410$, Table 2.1, Figure 2.3). On average, every additional year a regrowth forest will add one species (± 1.23).

Table 2.1 Simple linear regression results for landscape and transect characteristics *versus* species richness for 29 regrowth forest transects.

Variable name	<i>Coeffecient</i>	<i>Standard Error</i>	<i>P</i>	<i>R</i> ² _{adj}
Log ₍₁₀₎ nearest forest size	3.93992	1.565738	0.018100	0.159965
Nearest forest distance	-0.01632	0.021882	0.462301	0.016111
Log ₍₁₀₎ dist. to non-frag. forest	-8.43002	1.667841	0.000026	0.467148
Regrowth forest size	-0.01404	0.007602	0.075835	0.079217
Cont. agriculture plot size	-0.00717	0.002649	0.011631	0.184321
Age of regrowth forest	1.04883	0.231728	0.000109	0.410350

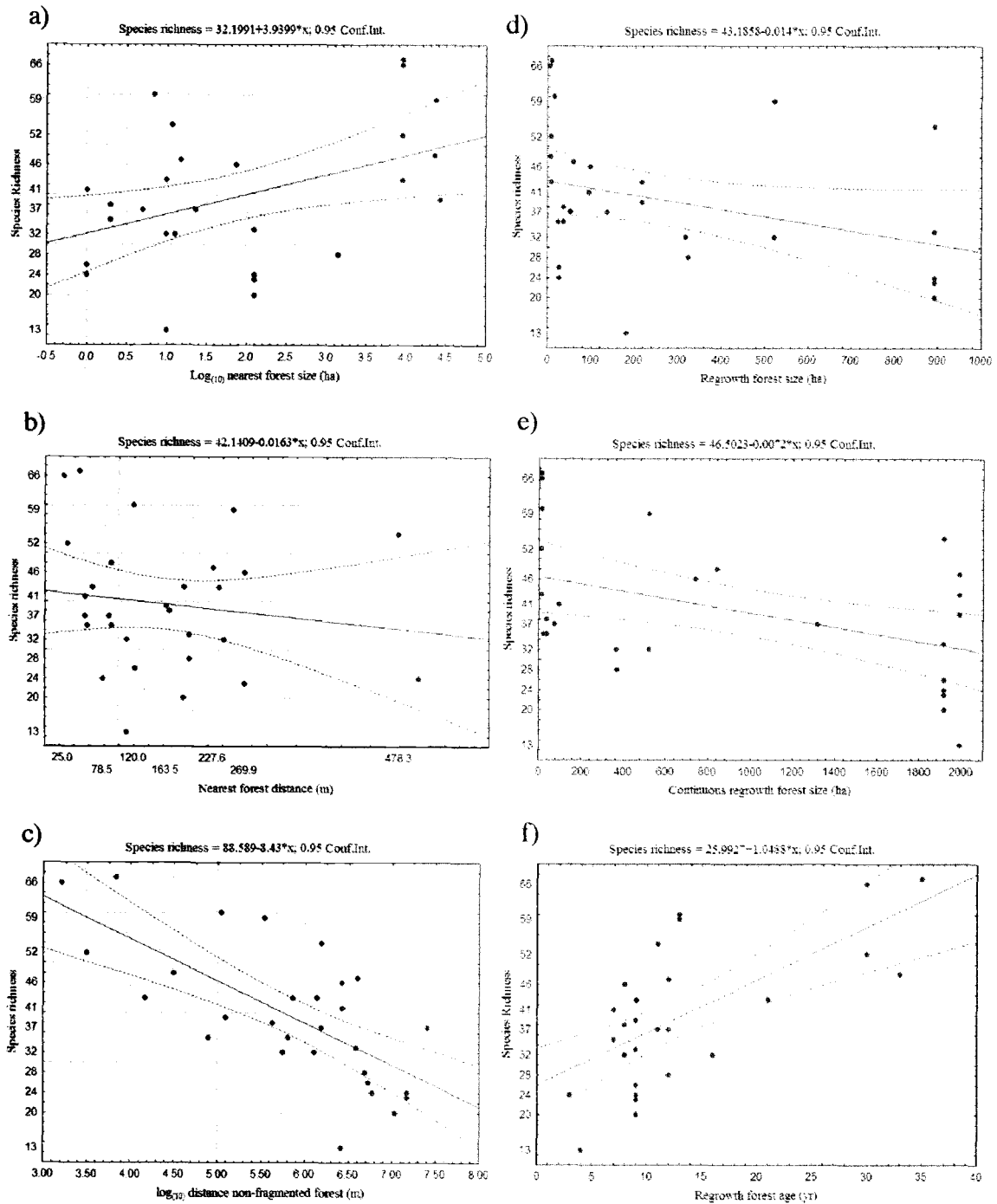


Figure 2.3 Scatterplots of simple linear regressions for the explanatory variable species richness *versus* independent variables of \log_{10} nearest forest size (a), nearest forest distance (b), \log_{10} distance to non-fragmented forest (c), regrowth forest plot size (d), continuous agricultural plot size (e) and age of regrowth forest (f). Solid red lines are the regression between the response and explanatory variables with the 95% confidence interval as a dashed red line. Blue dots are data points.

MULTIPLE LINEAR REGRESSION ANALYSIS

Stepwise forward regression analysis for the independent variables of $\log_{(10)}$ nearest forest size, nearest forest distance, $\log_{(10)}$ distance to non-fragmented forest, regrowth forest plot size, continuous agricultural plot size and regrowth forest age are used to explain their associations against the dependent variable of regrowth forest tree species richness. The stepwise forward procedure concluded in two steps (Table 2.2). From this regression analysis the only variable that was found to be significantly associated with regrowth tree species richness is $\log_{(10)}$ distance to non-fragmented forest ($p < 0.0001$, $R^2_{adj} = 0.467$, Table 2.2). Results for the regression model are identical to those reported for $\log_{(10)}$ distance to non-fragmented forest (Table 2.1).

The stepwise backwards procedure for the independent variables produced different findings relative to the results of stepwise forward procedures. The stepwise procedure sequentially removed continuous agriculture plot size, regrowth forest plot size and nearest forest distance (Table 2.3). Age of regrowth forest and distance to non-fragmented forest are highly significant in their association with regrowth forest tree species richness ($p < 0.0001$, $R^2_{adj} = 0.505$, Table 2.3 and Table 2.4). In this model, with each additional year a transect will gain 0.73 species (± 1.30) while for 100 m distance from non-fragmented forest a transect will lose 1.23 species (± 2.62) from an initial 37 species.

When an indicator variable is added to account for previous tenures that use fire, it was found not to have a significant influence on regrowth forest tree species richness after being corrected for distance to non-fragmented forest and regrowth forest age. In the regression model with $\log_{(10)}$ distance to non-fragmented forest and burning, burning

Table 2.2 Summary statistics of the stepwise forward multiple linear regression model for the effects of the independent variables, $\log_{(10)}$ nearest forest size, nearest forest distance, $\log_{(10)}$ distance to non-fragmented forest, regrowth forest size, continuous agriculture plot size and age of regrowth forest *versus* the dependent variable species richness.

Summary of stepwise regression; variable: Species richness

Forward stepwise

P to enter: 0.05, P to remove: 0.05

	Steps	Degr. of Freedom	F to remove	P to remove	F to enter	P to Enter	Effect status
Nearest forest distance	Step 1	1			0.55605	0.462301	Out
Regrowth forest size		1			3.40891	0.075835	Out
Cont. agriculture plot size		1			7.32723	0.011631	Out
Age of regrowth forest		1			20.48576	0.000109	Out
$\log_{(10)}$ Nearest forest size		1			6.33192	0.018106	Out
$\log_{(10)}$ Distance to non-frag. forest		1			25.54747	0.000026	Entered
$\log_{(10)}$ Distance to non-frag. forest	Step 2	1	25.54747	0.000026			In
Regrowth forest size		1			0.01081	0.917982	Out
Cont. agriculture plot size		1			0.11850	0.733434	Out
Age of regrowth forest		1			1.72419	0.200626	Out
$\log_{(10)}$ Nearest forest size		1			0.14577	0.705716	Out
Nearest forest distance		1			3.00624	0.094789	Out

Table 2.3 Summary statistics of the stepwise backward multiple linear regression for the effects of the independent variables, nearest forest size, nearest forest distance, distance to non-fragmented forest, regrowth forest size, continuous regrowth forest size and age of regrowth forest *versus* the dependent variable species richness.

Summary of stepwise regression; variable: Species richness

Backward stepwise

P to enter: 0.05, P to remove: 0.05

	Steps	Degr.Of Freedom	F to Enter	P to Enter	F to Exit	P to Exit	Effect status
Nearest forest Size	Step 1	1	0.08880	0.768506			Removed
Nearest forest distance		1	5.22839	0.032210			In
Distance to non-frag. forest		1	2.08574	0.162772			In
Regrowth forest size		1	2.18799	0.153268			In
Cont. agriculture plot size		1	0.83720	0.370120			In
Age of regrowth forest		1	8.30667	0.008653			In
Age of regrowth forest	Step 2	1	11.24157	0.002756			In
Nearest forest distance		1	5.40349	0.029283			In
Distance to non-frag. forest		1	3.00102	0.096603			In
Regrowth forest size		1	2.22447	0.149429			In
Cont. agriculture plot size		1	0.78124	0.385909			Removed
Nearest forest size		1			0.088799	0.768506	Out
Age of regrowth forest	Step 3	1	12.61488	0.001621			In
Nearest forest distance		1	4.79517	0.038496			In
Distance to non-frag. forest		1	4.78935	0.038604			In
Regrowth forest size		1	2.45118	0.130528			Removed
Cont. agriculture plot size		1			0.781238	0.385909	Out
Nearest forest size		1			0.001901	0.965603	Out
Age of regrowth forest	Step 4	1	10.87705	0.002919			In
Nearest forest distance		1	2.21928	0.148806			Removed
Distance to non-frag. forest		1	8.07989	0.008786			In
Regrowth forest size		1			2.451184	0.130528	Out
Cont. agriculture plot size		1			0.937764	0.342512	Out
Nearest forest size		1			0.008097	0.929048	Out
Age of regrowth forest	Step 5	1	8.94043	0.006029			In
Distance to non-frag. forest		1	6.17714	0.019698			In
Nearest forest distance		1			2.219280	0.148806	Out
Regrowth forest size		1			0.003576	0.952789	Out
Cont. agriculture plot size		1			0.115278	0.737050	Out
Nearest forest size		1			0.009971	0.921255	Out

Table 2.4 Stepwise backward final regression results for the effects of the independent variables distance to non-fragmented forest and age of regrowth forest *versus* the dependent variable species richness.

ANOVA					
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>
Regression	2	2808.353	1404.177	15.29548	0.000041
Residual	26	2386.888	91.80339		

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	36.51866	5.343806	6.83383	0.000000	25.53431	47.50301
Age	0.73717	0.246540	2.99006	0.006029	0.23040	1.24394
Distance to non-frag. forest	-0.01231	0.004955	-2.48538	0.019698	-0.02250	-0.00213

$R^2_{adj} = 0.505221$
n = 29

is not significant ($p = 0.528$, for the parameter estimate of burning). In the regression model of age of regrowth forest, distance to non-fragmented forest and burning versus the dependent variable of regrowth tree species richness, is not significant ($p = 0.518$). In both models the strength of the association is reduced as indicated by the R^2_{adj} values (Appendix A).

THE AGE OF REGROWTH FOREST, DISTANCE TO NON-FRAGMENTED FOREST AND SPECIES RICHNESS

Age of regrowth forest and distance to non-fragmented forest are consistently shown to be the most significant factors measured associated with regrowth tree species richness. Graphical procedures of the interaction between distance to non-fragmented forest, age of regrowth forest, and regrowth forest tree species richness are presented in three dimensional and two dimensional graphs with the z axis (species richness) represented by color coded isoclines (Figure 2.4). The graphing procedure shows there is a pronounced increase in tree species richness after 7 years. Distance to non-fragmented forest further increases in regrowth tree species richness at distances less than 600 m. Data points that are within 100 m of non-fragmented forests and older than 25 years have the highest recorded regrowth forest tree species richness.

FORESTS, FRAGMENTED FORESTS AND NON-FRAGMENTED FORESTS ASSOCIATION WITH REGROWTH TREE SPECIES RICHNESS

The percent of forest within 400 m of the regrowth forest transect was found to be significantly associated with regrowth forest tree species richness ($p < 0.002$, $R^2_{adj} = 0.352$, Table 2.5, Figure 2.5). On average, every 10 percent increase in forest cover will increase species richness 3.33 (± 4.36) beyond the initial 31 species. The proportion

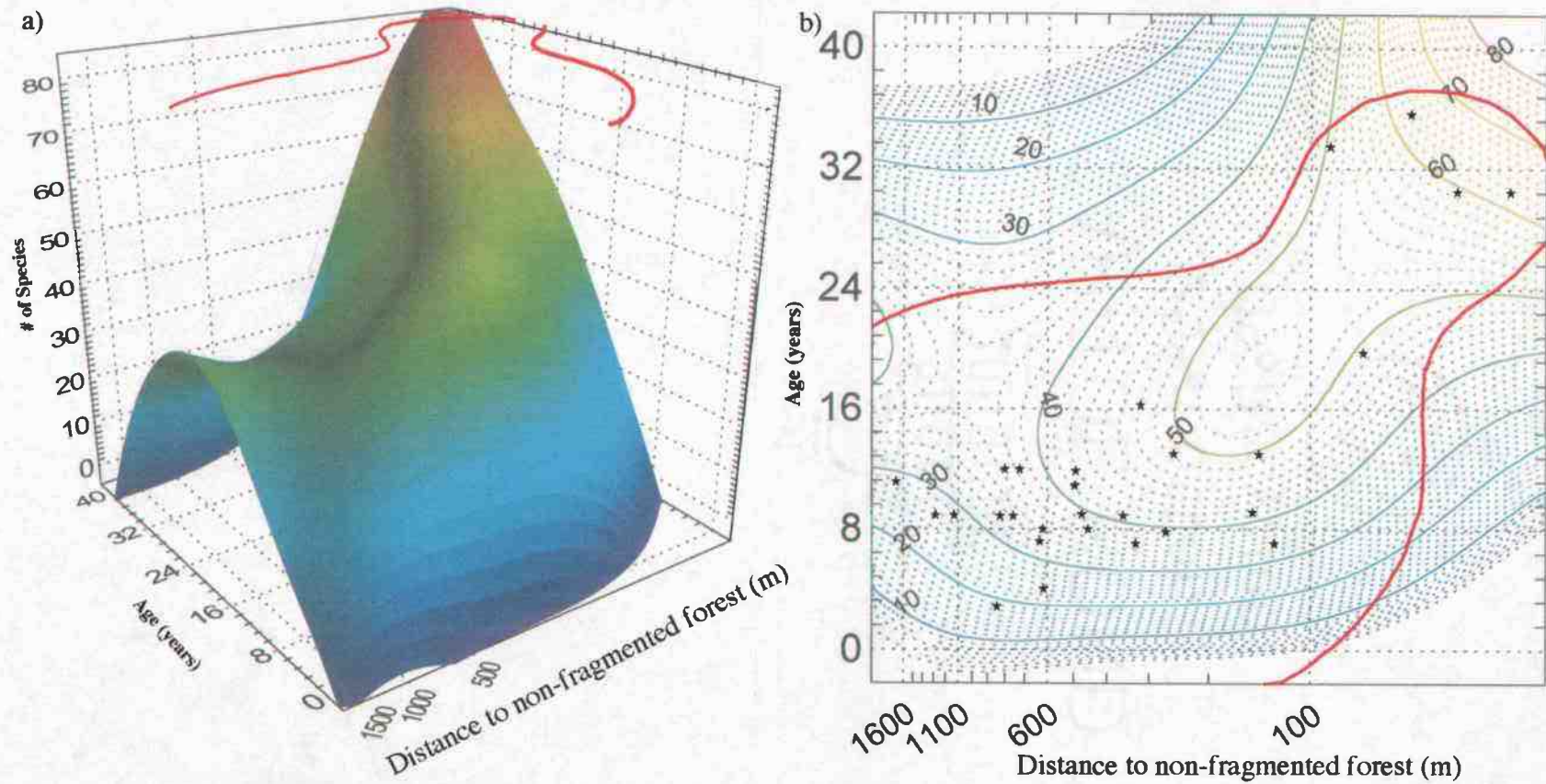


Figure 2.4 Quintic interpolation of the relationship between species richness, age of secondary forest and distance to the nearest continuous forest. Part a) is a three dimensional representation with the 95% confidence ellipsis of data points is shown as a red line over the interpolated model. Part b) is a two dimensional model with isoclines for the number of species present. The red line is the 95% confidence ellipsis of data points with black stars showing the location of the data points. Both graphs are different visual representations of the same quintic interpolating function using the transect data. The quintic interpolation methods is forced to use zero as a floor. As interpolated the influence of continuous forest seed dispersal elevates the species richness in areas that are within 600 m of the forest and that are greater than seven years of age.

Table 2.5 Simple linear regression results for percent of forest and non-fragmented forest *versus* species richness at 400 m for 23 regrowth forest transects.

Variable	Coeffecient	Standard Error	<i>P</i>	R^2_{adj}
% Forest w/in 400 m	33.26155	9.249495	0.001699	0.351635
% Frag. forest w/in 400 m	-216.601	56.81543	0.001017	0.380875
% Non-frag. forest w/in 400 m	33.46092	8.128479	0.000492	0.420223

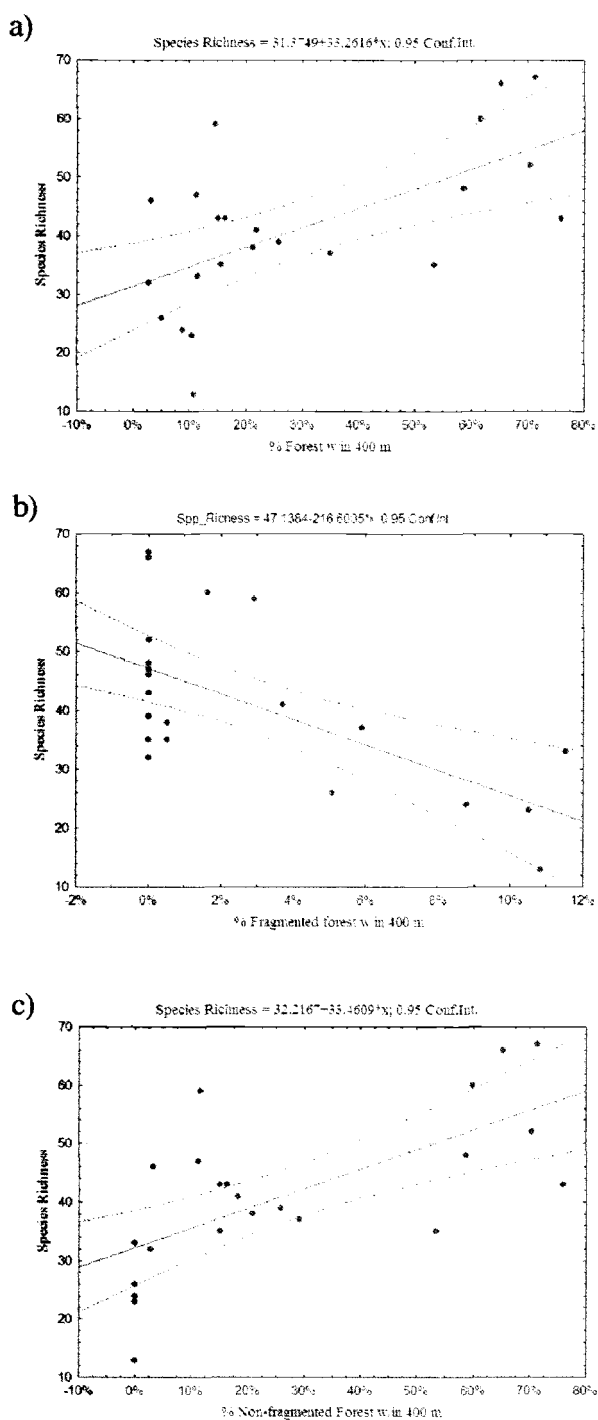


Figure 2.5 Single linear regression analysis for the dependent variable of (a) regrowth forest tree species richness versus percent forest within 400 m of the transect, (b) percent fragmented forest within 400 m of the transect, and (c) percent of non-fragmented forest within 400 m of the transect.

of fragmented forest's influence on regrowth forest tree species richness is negative and highly significant ($p = 0.001$, $R^2_{adj} = 0.381$). In this model, for every one percent increase in fragmented forest, there is a 2.16 (± 2.66) decrease in species from 47 species present. The percent of non-fragmented forest within 400 m of the regrowth forest transect is positive and highly significant in its association with regrowth forest tree species richness ($p < 0.001$, $R^2_{adj} = 0.420$). On average, a ten percent increase in non-fragmented forest yields 3.35 more species (± 3.81) above an initial 32 species.

DISCUSSION

The results of the single linear regressions confirm three factors that are significantly associated with regrowth forest tree species recruitment from large-scale disturbances, age of regrowth, size of the nearest forest, and distance to non-fragmented forest. These factors may be explained by regrowth forest tree species richness dependence upon proper microclimatic conditions, the availability of seeds and the ability of the seeds to disperse to establishment sites. The first factor is the age of the regrowth forest which is shown to have a highly significant association on tree species richness occurring in transects of regrowth forests. This is consistent with previous studies (Saldarriaga *et al.* 1988, Steininger 1996, Mesquita 1995, Honvák *et al.* 1995, Honvák *et al.* 1996, Mesquita *et al.* 2001, Steininger 2001, Lucas *et al.* 2002) and follows the recognized successional path in which microclimate plays an important role in successful establishment of saplings. In open areas, the extremes of temperature, humidity and wind shear are greater relative to closed canopy forested systems. These temperature extremes are seen as being too harsh for shade-tolerant forest species which can establish but rarely hold occupancy for longer than two years (Hubbell 1999). Pioneer species are adapted to these extremes with germination and establishment possible immediately after abandonment (Mesquita 1995, Steininger 1996, Williamson *et al.* 1998). Pioneer species are also the primary dominants for the first twenty years post-abandonment (Saldarriaga *et al.* 1988, Steininger 1996, Mesquita *et al.* 1999, Lucas *et al.* 2002).

As the regrowth forests age, species richness should increase as early dominating species such as *Cecropia spp.* or *Vismia spp.* cede to other tree species. These early secondary forests are typically dominated by the *Cecropia spp.* or *Vismia spp.* In

unburned secondary forest *Cecropia* spp. occupies a high percentage of the canopy for the first 30 years (Williamson *et al.* 1998, Mesquita *et al.* 2001, Lucas *et al.* 2002). Once canopy closure occurs, a richer assortment of tree species is capable of withstanding the buffered microclimatic regimes. After these species successfully establish, they will maintain a understory position until *Cecropia* loses its canopy dominance and canopy gaps form. These established understory species have the best advantage of gaining canopy occupancy (Saldarriaga *et al.* 1988). In regrowth forests that were previously burned, *Vismia* spp. dominates but does not dominate the canopy to the extent of *Cecropia* in unburned sites (Williamson *et al.* 1998, Lucas *et al.* 2002). *Vismia* spp. has a thicker crown and spreads by clonal growth. The long-term succession of *Vismia* forests has not been investigated or reviewed in the literature, presumably due to the fact that *Vismia* dominated forests are reburned before latter successional species turnover occurs (Bergen and Steininger, in Chapter Three).

Our study found that the size of the nearest forest has a significant association with the number of species that will recruit to adjacent regrowth forests (Tables 2.1). It is well established, especially within neotropical forests that as areas increase the number of species increase (Rankin-de Mérona *et al.* 1992, Condit *et al.* 1999). Species area relationships are more dramatic in alpha diverse systems where all species are rare, meaning that as area is increased the species richness will increase accordingly. Recent studies by Laurance *et al.* (2004) have documented that 88 % of all the tree species surveyed at BDFFP occur at densities of less than one individual per hectare. Assuming that forest area is proportional to the number of tree species that exist within these forests (Preston 1962, $S = cA^z$), this is expected. Larger forests will act as tree species

reservoirs, from which tree recruitments will occur once microclimatic extremes subside allowing for their successful germination and establishment.

Having a reservoir of mature forest tree species (*i.e.* forest size) and buffered microclimatic conditions (*i.e.* age) are not the only ingredients necessary for establishment to occur. Seeds need to be transported to places where they can establish within the secondary forests. Within the Amazon Basin, passive seed dispersal systems do occur but are not the predominant seed dispersal system. Passive seed dispersal systems represent less than 10% for trees within the Amazon (Reibero *et al.* 1999). Hubbell *et al.* (1999) noted that recruitment limitation is a major factor in limiting all species to establish in an available canopy gap. He further notes that 88% of the species at BCI have not had seeds deposited in passive seed traps over a ten-year period. Apparently, active animals that disperse seeds typically avoid depositing seeds within the traps. The majority of species seeds are dispersed actively with mammalian or avian vectors. Without these vectors to transport and deposit seeds, germination and establishment will be limited to sites beneath the parent where intraspecific competition between cohorts and with the parent tree all but exclude establishment proceeding beyond the initial years. The ultimate persistence of tree species with active seed dispersal is dependent on the persistence of the proper animal vectors (Terborgh 1988, Terborgh *et al.* 2001). We found $\log_{(10)}$ distance to non-fragmented forest to be significantly associated with regrowth forest tree species richness. The results of stepwise forward and backward regression analyses, and single regression analyses consistently suggest that the distance to these larger forests is more important than the distance to the nearest forest. The nearest forest distance does not have any association with tree species richness in

regrowth forests (Table 2.1). Two possible explanations exist for the differences between the larger and continuous forests and the nearest forest. The first could be that the nearest forest in the 29 transects taken were too small to contain species that were capable of establishment in buffered microclimatic extremes. The second is that larger forests contain seed dispersal vectors that are not found within the smaller and closer forests.

The factors influencing regrowth forest dynamics present a picture of the how regrowth forest tree species assemble. The age of the forest can be thought of as a surrogate for the vertical complexity of the forest, *e.g.* physiognomy. This vertical architecture affects the microclimatic conditions necessary for seedling establishment (Lovejoy *et al.* 1986). Nearest mature forest size is a surrogate for the number of species available to establish. The majority of the tree species require their seeds to be actively dispersed by active vectors, *e.g.* frugivorous mammals and birds. This is indicative of the highly significant association distance to these larger non-fragmented forests capable of sustaining seed vectors. Amazon forest dynamics are bio-complex, meaning that the majority of the tree species depend upon active dispersal of both their pollen and their seeds (Terborgh *et al.* 2001). In the simplest scenario, the persistence of one tree species is dependent also on the persistence of one suite of species to disperse its pollen and another different suite of species to disperse its fertilized seed. The distance to the nearest non-fragmented forest is in effect the probability of these seeds being transported to new establishment sites by their requisite vectors.

With regrowth plot size being non-significant, while continuous agriculture plot size is significantly negative associated with tree species richness, we interpret these findings to support our observations on distance to non-fragmented forest. Regrowth plot

sizes are smaller, and exist within the continuous agricultural plot matrix bordered by mature forests. Larger continuous agricultural plots are also spatially associated with greater distances to non-fragmented forests (all other things being equal). We would expect there to be a decrease in regrowth forest tree species richness.

The results from the stepwise linear regression analyses shed light on the interaction between the three factors that have significant association with tree species richness in regrowth forests discussed above. The stepwise forward regression shows the strongest effect is the $\log_{(10)}$ distance to non-fragmented forest (Table 2.2). The stepwise backwards regression analyses conclude that age and the distance to non-fragmented forest are the strongest associations of the landscape and regrowth forest characteristics measured (Table 2.3). The stepwise backwards regression results are stronger in their capacity to explain the variance associated with regrowth forest tree species richness but only by four percent. This ranking has not been shown elsewhere in studies of Amazon regrowth forests and increases our understanding of the importance of very large, non-fragmented forests that contain a full complement of species assemblages. This finding also supports the idea that smaller forests do not contain all the necessary dispersal species required to provide for their persistence through time. Also, smaller fragmented forests are losing their tree species richness as they are invaded by pioneer species (Laurance *et al.* 1997, Laurance *et al.* 1998) and when tree fall gaps are created by large shade-tolerant deep forest species (Laurance *et al.* 2000). The biological collapse of species contained within forest fragments is going to limit the availability of species that can recruit to new sites. This study shows that tree species richness decreases when the

percent of fragmented forests increases within 400 m of the transects, thus fragmentation at this scale will have an impact on forest diversity.

The effects of previous land tenure are not significantly associated with regrowth forest tree species richness when corrected for $\log_{(10)}$ distance to non-fragmented forest or distance to non-fragmented forest with age of regrowth forest. Previous studies have noted that previous land tenure will affect biomass sequestration (Uhl *et al.* 1988, Steininger 1996, Steininger 2001) but seems to have little influence on subsequent regrowth forest tree species richness. Since burning changes the successional trajectory of regrowth forest, with unburned forests being dominated by *Cecropia spp.* and burned forest dominated by clonal *Vismia spp.*, (Williamson *et al.* 1998, Mesquita *et al.* 1999), one would assume that tree species recruitment also differs in these regrowth forest types. Our study does not support this. It is more likely that the establishment of vertical architecture is sufficient to establish other species that will eventually gain canopy occupancy and successional dominance.

Our graphical evidence of the interaction between distance to non-fragmented forest, regrowth forest age, and regrowth forest tree species richness shows that areas within 600 m of continuous forest and that are older than seven years of age have an elevated richness. This species enrichment is more prominent in areas that are older and closer to the continuous forest edge. It is most likely that older forests would also have an elevated species richness but regrowth forests that are older than 20 years of age are rare (Bergen and Steininger, in Chapter Three) and underrepresented in the data transects (as delineated by the 95% confidence ellipse based on data point locations). One reason for the rarity of secondary forests in this age is that by year 20, soils have regained

nutrient levels that allow for agricultural crops or pasture. Considering that it requires much more physical labor to clear mature forests relative to secondary forests, older secondary forest are of higher value and are sought to convert back to active agriculture (Bergen and Steininger, in Chapter Three.). If there is high population pressure or if older secondary forests are in short supply, then further deforestation is more likely to occur in areas that are closer to active economic markets and reliable transportation. Within this study area, regrowth forests have a higher turnover rate if they are closer to metropolitan Manaus compared to more remote northern locations (Bergen and Steininger, in Chapter Three). Decreased turnover rates mean that the average secondary forest is of a younger age and thus less capable of providing establishment sites for most mature forest species.

FRAGMENTED FORESTS *VERSUS* NON-FRAGMENTED FOREST

The results of the analysis of forest composition within 400 m of the transects signifies the differences between fragmented and non-fragmented forests and their association with proximal regrowth forest's tree species richness. Non-fragmented forest percentage is positively associated with regrowth forest tree species richness, meaning that the more non-fragmented forest occurring within 400 m of regrowth forest the higher the tree species richness will be. The percentage of fragmented forest is negatively associated with tree species richness within 400 m, meaning that as you increase the composition of fragmented forests tree species richness will decline in regrowth forest areas. These results support previous discussions that fragmented forests are less capable of providing a species diverse seed source and also lack the requisite animal dispersal

vectors. Further experiments capable of decoupling the effects of seed diversity and animal dispersal within fragmented forest are needed in order to assign relative importance among and between these two factors. It could also be true that the two factors are inter-dependent and can not be decoupled within natural environments as would be the case if a tree species were required to have an obligate animal dispersal for establishment via gut passage with the animal vector being strictly dependent upon the fruit for food. Excluding either the fruit or the animal vector will discourage the longevity of the other. Similar trophic cascades have been recognized on BCI as well as recently isolated islands in Venezuela (Terborgh 1988, Terborgh *et al.* 2001).

IMPLICATIONS

This study provides a clearer understanding of how secondary forests naturally succeed into mature forests if they are given time, have the necessary seed source, and the required dispersal vectors. Using the linear regressions calculated for the age of secondary forest, it will take 242 years for secondary forest to regain the full tree species richness that it had previously had (Oliveira & Mori, 1999, we used the conservative 280 for this calculation). This estimate must be interpreted with caution but is within the range of that presented for biomass by Saldarriaga *et al.* (1988). Non-fragmented forests are necessary for the rehabilitation of tree species richness in previously anthropogenically altered areas because of their seed bank and providing habitat for animal vectors.

Non-fragmented forests have a more diverse seed bank for tree species richness. Ultimately this seed bank will determine how many species are available to recruit to

potential establishment sites. The larger the non-fragmented forest is, the more tree species it will contain. Small forests will contain fewer potential recruits (but this can be optimistic). If a forest is too small, it will not contain the species necessary to complete pollination because either the pollination vectors (typically insects, birds and volant mammals) or the gamete from another individual is too far away. Within the central Amazon 88% of the species identified occur at densities lower than 1 individual ha⁻¹ (Laurance *et al.* 2004). Amazonian forests have their tree species spatially arranged in what is described as an alpha-diverse system (Whitaker 1977). Alpha diverse systems have many species that are not segregated by 'beta' community types within a gamma diverse landscape matrix (Whitaker 1977) but all species occur rarely across the region. One implication of this spatial arrangement of diversity is that most species are always rare and most areas contain high species diversity. It can be stated that for mature forests less than 10 ha in size, the potential of producing fertile seed for the majority of self-incompatible tree species is severely inhibited. This is consistent with the findings that mature forest tree species recruitment is significantly lower in edge dominated, small area forest fragments (Laurance *et al.* 1997, Laurance *et al.* 1998). Laurance *et al.* (1998) attributed the prevalence of early successional species to changes in micro-climate, but the lack of available seed and seed dispersed by animal vectors is equally plausible. Forest fragment edges are surrounded by secondary forests that produce copious amounts of fruiting bodies that are easily transported past adjacent forest edges actively or passively.

Simply having seeds in the bank does not guarantee that these forests will have the necessary vectors to disperse them. Large bodied mammals and large ground birds

are typically the first species to be killed by hunters within the Amazon (Robinson *et al.* 1999, Laurance 2001b, Fa *et al.* 2002, Peres and Lake 2003). These species also play an important role in dispersing the fruits that they consume to new establishment areas.

When smaller forests, or even larger continuous forests are accessible by hunters, they are devoid of larger bodied vertebrates due to increased depredation (Robinson *et al.* 1999). Continuous forests can be considered to be reserve banks of the vectors that disperse seeds into adjacent regrowth areas. Many interior forest species will avoid younger regrowth forests. Many primate species prefer very old regrowth forests because of the increased availability of fruiting trees. Non-fragmented forests are more able to be re-colonized once they have been depredated since species dispersal is not inhibited by foreign habitat barriers blocking animal dispersal (Laurance 2001b).

Current development trends throughout the Brazilian Amazon Basin are intensifying (Laurance *et al.* 2001, Nepstad *et al.* 2002, Carvalho *et al.* 2001). The majority of deforestation that occurs within the Brazilian Amazon is along the South to Southwest 'arc of destruction.' Within this forested area, cattle farms conserve 50% of their land as forest although this varies depending on the total amount of acreage of the property site. Considering that all but the largest *fazendas* exceed 10,000 ha, this formula creates forest reserves that are too small to contain viable populations of tree species and the vectors needed to disperse both pollen and seed. These reserve forests are also created with no spatial orientation to existing forest tracts. Ultimately, a patchwork of forested fragments and fire maintained agricultural land is created. These smaller forests are also in extreme danger of being impacted by escape agricultural fires (Nepstad *et al.* 2002, Cochrane *et al.* 1999, Netstad *et al.* 1998). The dynamics of the policy, although

appearing to be conserving forests, do not match the ecological criteria necessary to sustain the forest or to rehabilitate fallow agricultural lands. In the long term, these conservation practices have slowed deforestation by promoting the long-term decay within forested fragments not capable of sustaining themselves. If these conservation strategies were changed so that large tracts of continuous protected forests were established via mitigation banks, the longevity of these forest systems would be increased greatly because these forests have the potential to contain the ecological functions needed for forest persistence (protection meaning that hunting would be limited and there would be agricultural management of adjacent farmlands to inhibit the spread of escape management fires). In areas such as the states of Mato Grosso and Southern Para, where agricultural development pressures are increasing, such a change may increase the likelihood of saving the remaining biological species diversity relative to the troubled and controversial conservation procedures presently in place.

With the continued progress of several large-scale infrastructural development projects (*Pluri-annual 2004-2007*), the deleterious effects of these projects could be curtailed by insuring that large continuous tracts of forest are not fragmented further by concentrating linear projects such as railroads, road improvement and utility corridors into spatially discrete linear complexes, *e.g.* a railroad adjacent to an improved highway adjacent to a utility corridor (Bergen *et al.*, in Chapter Four). The increase of infrastructural development will locally be more pronounced but the surrounding forest is more capable of persisting with ecological integrity and function since it is not fragmented. Additively, if changes in infrastructure were developed for *Pluri-Annual 2004-2007*, the amount of forest saved could be considered for Clean Development

Mechanisms provided for carbon trading under the current standards of the most recent accords of the Kyoto Protocol. These types of changes have yet to be capitalized by the Brazilian Federal Government, despite the fact it has pledged monies for environmental impact studies for *Avança Brasil* (*Avança Brasil* was the initial federal program that has recently been renamed as Pluri-annual 2004 - 2007). The Brazilian Federal Government has neither accepted nor rejected the Kyoto Protocol, further confining the longevity and security of such management scenarios to the realm of academic conjecture. The inability of the Brazilian Federal Government to come to a decision on these important political issues, only insures the advancement of management practices to conserve natural resources that are not effective. Ultimately, the failure of these management practices will insure further future international scrutiny.

CONCLUSION

Regrowth forest's tree species richness is dependent on two primary factors, the architecture of the forest and the proximity to seed source. Once seven years has past a higher number of tree species can establish due to dampened micro-climatic variables. A viable seed source is not just dependent on an adjacent forest being able to provide seed but is also dependent on proper dispersal of these seeds. Stepwise multiple linear regression analysis has shown that the distance from a non-fragmented forest and the age of regrowth forest are the most significant factors associated with tree species richness in regrowth forests. These results show the importance of non-fragmented forests for they contain the full biologic integrity necessary for the persistence of these highly diverse forest systems. These ecological necessities dictate the sustainability of forest practices throughout the forested Amazon. By not allowing forest clearings to be wider than 500 m, regrowth forests will be able to rehabilitate the tree species diversity at an increased rate. Failure to recognize this through effective policies in landuse management not only will ultimately imperil rain forest integrity but also bring more international pressure upon the Brazilian state and federal government to change management practices that are not based upon ecological reason.

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Deforestation, Land Tenure, Fire and Inter-annual Variation's Effects on the Carbon Budget of a Central Amazon City

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ABSTRACT

The Amazon Basin is being studied for its contribution to global atmospheric gases exchanges, notably carbon dioxide (CO_2). CO_2 is stored in tropical rain forest biomass (carbon * 2.105). When rain forests are harvested, the stored carbombiomass is released to the atmosphere through ignition and subsequent decomposition. Forest plots are converted to agriculture use. These plots are then either burned for pasture maintenance or abandoned. The abandoned plots succeed into secondary forests that sequester carbon from the atmosphere thereby forming atmospheric carbon sinks. Thus, at any point in time forests are being deforested, agriculture and pasture plots are being maintained and other previous agriculture areas are abandoned. Because of the difficulty in determining secondary forest biomass with satellite imagery, estimations of secondary forest biomass across dynamic landscapes have remained approximate. We studied three transects centered on the city of Manaus ($60^\circ 00' \text{W}$, $2^\circ 30' \text{S}$) and tracked the influence of secondary forest dynamics relative to mature forest deforestation through a nineteen-year period. This spatially explicit model accounted for mature forest ignition and decomposition, and secondary forest sequestration, ignition and decomposition. We show that including regrowth forest dynamics in the measurement of total above-ground biomass (TAGB), significantly changes the estimates of TAGB. The methodology developed within this spatially and temporally explicit model is capable of discerning carbon sink habitats from carbon sources. One transect is estimated to be sequestering carbon through time while the majority of the area studied is losing TAGB. Frequent agricultural land turnover and high deforestation rates are associated with transects declining of TAGB. Land use characteristics change over time at varying rates and

change differently at different locations. The ability to model these disparate changes contributes to a better understanding of how land tenure affects TAGB through time while improving the accuracy of TAGB estimates from remotely sensed data.

INTRODUCTION

It has been of recent scientific and political interest whether the Amazon forests act as atmospheric carbon sources or sinks to global atmospheric carbon dioxide concentrations (Fernside 1995, Prentice and Lloyd 1997, Laurance *et al.* 1998, Houghton *et al.* 1999, Laurance *et al.* 2001, Steininger *in review.*). Areas that are increasing in total above-ground biomass are considered carbon sinks because they are continuously sequestering carbon dioxide (CO₂) from the atmosphere by incorporating it into biomass. Areas that are decreasing in total above-ground biomass (TAGB) are referred to as sources, meaning that they release biologically sequestered CO₂ into the atmosphere via combustion and decomposition. Scientific inquiry into the identification of carbon source and sink areas is important because of the potential large influence of the Amazon Basin on global bio-geo chemical cycles, global CO₂ concentrations in particular.

Models describing carbon cycles are diverse and intricate. The majority of studies identify three main factors in estimating carbon budgets over large regions:

- 1) immediate loss of carbon due to deforestation with subsequent combustion of mature forest biomass.
- 2) continued loss of carbon from decomposing biomass retained after combustion.
- 3) carbon sequestered from regrowth forests.

Carbon and biomass are both used to measure these carbon cycles. Carbon and biomass can be used interchangeably and have the relationship of 1 unit carbon equaling 2.105 units of biomass (Brown & Lugo 1990, Brown 1990). Most studies prefer to use the standard 1: 2 approximation.

Mature forested systems are considered to be at biomass equilibrium when carbon lost due to individual tree death is balanced by the carbon gained by more rapid tree growth occurring in the newly formed tree fall gap. Phillips *et al.* (1998) proposed that mature forests act as carbon sinks within the Amazon. Phillips *et al.* (2002) have documented increases in liana biomass from censuses of tree plot stands on the periphery of the forested Amazon Basin. Their findings suggest that tropical forest systems are increasing in biomass as a result of increased concentrations in CO₂. Other recent studies indicate that mature forest systems could be in species transition due to increases in atmospheric CO₂ concentrations (Laurance *et al.* 2004).

Recent models have shown that anthropogenic development in the Amazon could potentially be responsible for up to 14% of the total increase in global atmospheric CO₂ concentrations (Laurance *et al.* 2001) but this study ignored regrowth forest sequestration. Annual “bookkeeping” models have narrowed the contribution to 0.2 ± 0.4 Pg C yr⁻¹ (Houghton *et al.* 1999). The large inter-annual variance of ± 0.4 Pg C yr⁻¹ is to compensate for the effect of land tenure and forest fires. A more comprehensive estimate of 0.261 Pg C yr⁻¹ was calculated by Fearnside (1997). Defries *et al.* (2002) have addressed the need for greater precision in estimating land use dynamics within tropically forested areas. An increase in the accuracy of the contributions from tropical rain forests to global atmospheric processes requires a more detailed understanding of these tenure systems. Despite the inherent complexities, none of the models has accounted for the dynamics of regrowth forest tenure and rotation adequately. For example, Houghton *et al.* (1999) have applied to their ten-year model the ratio of pasture to secondary forest from measurements obtained from only one year. Single ‘snap-shot’ estimates of

temporally continuous processes have considerable imprecision and inaccuracy. If these initial errors are applied across several years, the error increases multiplicatively.

The goal of the current study is to use spatially explicit models of regrowth forest dynamics in measuring carbon sequestration and quantifying regrowth forest's contribution to carbon emissions and sequestration through time in a dynamic landscape in the central Amazon Basin.

REGROWTH FORESTS OF THE CENTRAL AMAZON

There have been several studies modeling regrowth forest biomass sequestration within the Amazon Basin (Uhl *et al.* 1988, Saldarriaga *et al.*, 1988, Nepstad *et al.* 1991, Brown & Lugo 1990, Brown and Lugo 1992, Fearnside 1992, Lucas *et al.* 1993, Fearnside & Guimarães 1996, Honzak *et al.* 1996, Steininger 1996, Lucas *et al.* 1998, Steininger 2001, Mesquita *et al.* 2001). Uhl *et al.* (1988) studied the potential of forest sequestration for three classes of regrowth forest and found that regrowth forest's sequestration of biomass differs according to the amplitude of the previous land management. In tenure systems where the topsoil was mechanically manipulated by tractor, biomass sequestration was delayed and occurred at 1/18th the rates of areas that were slashed and burned once. Brown and Lugo (1990) developed allometric growth equations that are widely used to calculate tropical regrowth forest biomass. Saldarriaga *et al.* (1988) estimated regrowth forest sequestration of forests near the intersection of Brazil, Columbia and Venezuela. These Guyana Shield *terra firme* regrowth forests were 10 to 80 years of age. Their study found that biomass accumulates at a linear rate until the stand is 40 years old at which point stand biomass sequesters more slowly. They

attribute the slower sequestration to individual tree death and gap replacement by later successional species. Biomass continues to sequester past 80 years through the combination of accumulated dead woody debris and large-diameter tree biomass.

Regrowth forest biomass sequestration has received attention within the central Amazon Basin in the vicinity of Manaus (2°45'S, 60° W). Honzák *et al.* (1996) quantified regrowth forest biomass relative to leaf area index (LAI) and to photosynthetically active radiation (PAR) within regrowth forests occurring 20 to 80 km North of Manaus on Guyana Shield terra firme forests. Using allometric equations developed by Brown and Lugo (1990), a linear growth model was used to produce a linear regression in which age would predict biomass. A rate of 10 Mg biomass ha⁻¹ yr⁻¹ was used to estimate plots two to 15 years of age. Beyond the 15 years of age the linear model overestimated total above ground biomass. Honzák *et al.* (1996) mentioned the greater variability associated with biomass estimation of regrowth forest of greater than 20 years of age. Lucas *et al.* (1998) estimated the regrowth sequestration as 8 Mg ha⁻¹ yr⁻¹ on former pasture land. Within regrowth forests less than 1 ha, sequestration of biomass proceeds at a rate of 11 – 15 Mg biomass ha⁻¹ yr⁻¹ until year 15. Steininger (2001) further compared tropical regrowth forests occurring in areas 60 km north and south of Manaus as well as regrowth forests occurring near Santa Cruz, Bolivia. His study found no significant difference between the regrowth forests that are north and south of the Amazon River. He calculated that 2 – 12 year old forests accumulate biomass at a near linear rate with the highest variability occurring in the youngest stands (< 4 years of age). Regrowth sequestration is temporarily hindered in stands that were formerly maintained as pasture, at a biomass sequestration rate of 5.0 Mg ha⁻¹ yr⁻¹.

Within the 2- 12 year age class forests sequestered biomass at the rate of 9.1 Mg biomass ha⁻¹ yr⁻¹. After 12 years of age, regrowth forest biomass sequestration slowed to a mean of 5.9 Mg ha⁻¹ yr⁻¹.

MATURE FORESTS OF THE CENTRAL AMAZON

Biomass of mature forests of the central Amazon have been described for species composition (Riebhiero *et al.* 1999, Rankin de Morona *et al.* 1990, Rankin de Morona *et al.* 1992). A number of studies have estimated mature forest biomass of forest types prevalent near Manaus. Saldarriaga *et al.* (1988) estimated that mature forest biomass is comprised of between 223 and 271 Mg ha⁻¹ in four Guyana Shield terra firme forest sites approximately 900 km north of Manaus. Lucas *et al.* (1998) estimated mature forest biomass based on five previous estimates performed by studies within 90 km of Manaus. These estimates averaged mature forest biomass to be 362 Mg ha⁻¹ within a range of 268 to 483 Mg ha⁻¹. Forest structure, biomass and composition have also been estimated relative to soil and environmental site conditions (Laurance *et al.* 1999). Laurance *et al.* (1999) estimated the average mature forest as 356 ± 47 Mg biomass ha⁻¹. This measurement was based on 64 one ha tree plots from the Biological Dynamics of Fragmented Forests Project (BDFFP). Although there is variability in forest structure, composition and biomass, we feel the consistency of reported biomass estimates allow for accurate estimation over broad spatial scales. Supporting this assertion is the fact that the spatial extent of this study is within 40 km of all the previously mentioned studies occurring in the Manaus area. Although edaphic and topologic factors have been found to affect total above-ground biomass quantitatively, the techniques to identify these areas

from MSS, TM and ETM+ satellite data have not developed to a point where these estimates can be verified by ground measurements. The inability to identify forest biomass heterogeneity from satellite imagery has restrained large-scale fine resolution studies to a primary forest/ non-forest or a forest/ secondary forest/ pasture classification scheme (Skole and Tucker 1993, Houghton *et al.* 1999, respectively, but see Killeen *et al.* 1998 in Bolivia, Lucas *et al.* 1993 and Steininger 2001 for studies that use more complicated habitat classifications with ground verification).

THE FATE OF DEFORESTED FOREST BIOMASS

The process in which deforested carbon is released to the atmosphere has been studied and replicated within the Manaus region (Fernside 1992, Fernside *et al.* 1993 and Carvalho *et al.* 1995). Fernside (1992) reported that above-ground biomass volatilized by combustion to the atmosphere is 28.4%. Decomposition accounts for 69.0% with 2.6% remaining onsite after ten years. After three burnings 35.0% of the original biomass is released into the atmosphere and 61.9% is released by decay. Fernside *et al.* (1993) subsequently reported 27.4% combustion efficiency in a forest cleared for pasture north of Manaus. Carvalho *et al.* (1995) produced a detailed study of the actual structure and composition of forests before and after combustion. He reported a combustion efficiency of 25.1%. Decay percentages were assumed to equal residual non-combusted biomass. With the exception of Fernside's (1992) 69% decomposition occurring over a 10-year period, decay processes are less reported. Decay functions are negatively exponential with large amounts of carbon given off after combustion, then decreasing exponentially until after 10 years only negligible amounts are released. Negative exponential functions

have recently been used to quantify the contribution of deforested mature forest decay through time (Houghton *et al.* 1999, Steininger *in review*). Regrowth reforest combustion efficiency is likely to be much higher (Steininger *in review*) since the surface area to volume ratios of regrowth forest structure are high relative to the more massive mature forest structure. Regrowth forests are typically composed of low-density wood that is easily volatilized. Regrowth combustion efficiencies of 60% can be expected.

MODELING REGROWTH FORESTS IN THE CENTRAL AMAZON

Multi-spectral Thematic Mapper imagery (TM) to estimate regrowth forest age has previously been used within the study region (Lucas *et al.* 1993). In the time series between 1976 to 1991 (1976, 1977, 1985, 1988, 1989 and 1991), each annual Multi-Spectral Scanner (MSS) or TM image was geo-referenced within one pixel width relative to all other satellite imagery. Each image was then classified according to pasture, secondary forest and mature forest classes. In areas that were classified as regrowth that also had subsequent years classified as regrowth, the authors added an annual increment to their age. Since an annual satellite imagery sequence was not used, the age added to a regrowth area was equal to the number of years between consecutive imagery dates. Areas that were regrowth and then subsequently classified as pasture had their age returned to 0. Other scientists looking at regrowth age *versus* biomass correlations, argue that age is a more accurate method to estimate biomass (Nepstad *et al.* 1991). Satellite imagery classification has not proven capable of identifying tropical regrowth biomass accurately. Honzak *et al.* (1995) produced accurate biomass estimates based on age mapping methodology within portions of the area studied by Lucas *et al.*'s (1993). Lucas

et al. (1996) advanced using age data to estimate regrowth forest's above-ground biomass. However, Lucas *et al.* (1996) did not account for mature forest biomass. Because no estimation of total above-ground biomass was conducted, carbon emissions from deforested mature forest biomass could not be calculated.

RECENT ESTIMATES OF AMAZON FOREST CARBON PROCESSES

A recent study of the Brazilian Legal Amazon's contribution to global carbon cycles has "modeled" three processes necessary to estimate carbon emission and sequestration throughout the basin in spatially explicit fashion (Houghton *et al.* 1999). This study incorporated mature forest biomass estimates from different pan-Amazon studies. It measured deforestation rates by comparing two classified TM image mosaics for 1986 and 1992. Deforestation rates were estimated via land use/ land cover change detection techniques. From deforestation statistics provided by INPE (the Brazilian National Space Agency responsible for tracking deforestation within the Brazilian Legal Amazon), they extrapolated annual deforestation rates to 1998. Relative to satellite-derived estimates of deforestation, notable overestimation was associated with INPE statistics (the over-estimate of deforestation by INPE has also been noted in other previous studies, Skole and Tucker 1993). With these "calibrated" deforestation rates they were able to estimate the annual contribution of mature forest carbon to combustion and decomposition. They were able to estimate regrowth forest sequestration and accommodate for the fraction of agricultural land in crops and pasture based on the land use composition in 1986. They compared the mature forest deforestation emissions to the amount of sequestration by regrowth forest and quantified an annual 0.2 Pg C

contribution to the global atmospheric carbon pool. Finally, Nepstad *et al.* (1999) speculate that fire, logging and regrowth forest rotation are not adequately accounted for and could possibly contribute as much as 0.4 Pg C annually to the global atmospheric carbon pool.

OBJECTIVES OF THIS STUDY

The purpose of our study is to quantify and then qualify secondary forest dynamics, fire, and logging occurring at scales larger than 30x 30 m within an area where considerable understanding of ground level carbon processes already exists. The carbon processes of mature forest deforestation and secondary forest dynamics including fire and decay have not been previously quantified relative to one another within one study. This study will;

- 1) quantify deforestation on an annual basis (1985- 1999) from spatially explicit high resolution satellite data.
- 2) quantify regrowth forest age at 30x 30 m resolution for the entire 14 year period.
- 3) estimate regrowth forest biomass by age-biomass allometric equations at annual increments.
- 4) estimate total above-ground biomass as the sum of mature forest biomass and regrowth forest biomass annually.
- 5) estimate the carbon emission of mature forest combustion, mature forest decay, regrowth forest combustion and regrowth forest decay on an annual basis.

- 6) characterize anthropogenic land use patterns to better understand how land use patterns affect the metrics of total above-ground biomass, mature forest biomass, regrowth forest carbon sequestration and carbon emissions.

The estimates generated from our study will be used to produce more accurate carbon budget estimates by including data gained from the quantification and characterization of secondary forest dynamics and land tenure systems. The contribution of these areas to carbon emissions and biomass sequestration is important not only to the forests surrounding Manaus, but could apply to all pan-Amazon carbon budget studies.

MATERIALS & METHODS

STUDY AREA

Manaus is a central Amazon city of over two million people located at 2°45' South Latitude, 60°00' West Longitude. This area receives over 2,400 mm rainfall annually with moderately pronounced seasonal fluctuations. Regions north of the Amazon and Rio Negro Rivers are of Precambrian Guyana Shield origin (argilosols). Areas to the south of these rivers are typically of more recent alluvial origin (arenosol). Topographic relief increases moving north towards the center of the Guyana Shield. Relief can be locally steep, especially adjacent to stream systems, but typically does not vary by more than 80 m. Such relief is not prevalent south of the Rio Negro/ Amazon. Forest structure also varies along this elevation gradient and is consistent with the descriptions of *floresta de platô*, *vertente*, *campinarana* and *baixo* (Ribiero *et al.* 1999). Forests in this area are relatively well studied in comparison to other parts of the Amazon Basin. Detailed forest surveys of mature and regrowth forest types have occurred across the study area (Rankin de Merona *et al.* 1990, Rankin de Merona *et al.* 1992, Laurance *et al.* 1999, Ribiero *et al.* 1999, Mesquita 1995, Lucas *et al.* 1996, Honzák *et al.* 1995, Steininger 2001).

Initially, the goal of our study was to quantify and qualify changes occurring in the immediate vicinity of the Biological Dynamics of Fragmented Forests Project (BDFFP) as delineated by a 10 km buffer around BDFFP landholdings. A rectangle that enclosed the 10 km buffer area defined Transect 1. All spatial data are stored in a Universal Transverse Mercator coordinate system allowing for length and area measurements. Transect 1 is 61,440 x 36,900 m, comprising an area of 2,267 km². After

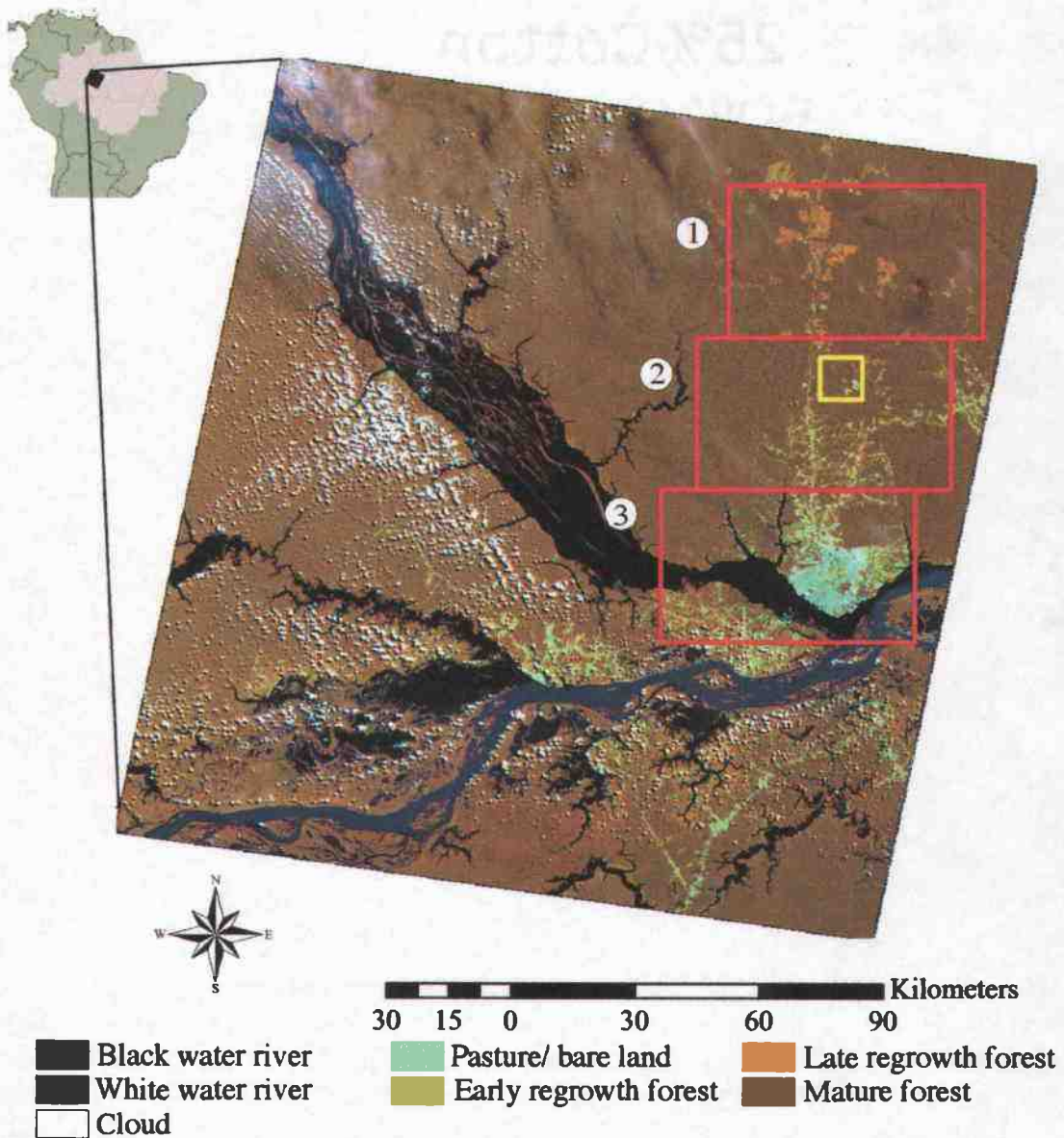


Figure 3.1 Locations of the three transects (red boxes) within the extent of 1997 TM imagery (bands 4, 5, 3 = RGB). Each transect is 61,440 by 36,900 m, an area of 2,267 km². Transect 1 occurs in the north. Transect 3 occurs furthest south. The Transect 2 subsection is used to display the spatially explicit behavior of the aging algorithm through time at high resolution (yellow box, 10,217 x 10,217 m).

successful model development within Transect 1, two more transects, each identical in size and shape were created to the south-southwest (Figure 3.1). The transect areas were shifted to the west in order to maintain the transects area despite the 231-62 (row-column) south-southwest boundary of the Landsat TM and ETM+ imagery. There is a pronounced decreased human population density gradient from the southern metropolitan Manaus to northern BDFFP *faviendas* and field study sites (*faviendas* are large cattle ranches).

COMPUTER AND SOFTWARE

Annual Landsat TM imagery was collected in full and 1/4 scene for the years 1985 to 1997. Landsat ETM+ imagery was acquired for the 1999 satellite imagery. All analysis of the satellite imagery was conducted on a Silicon Graphix Indigo 2 extreme workstation running the IRIX 6.5.7 operating system using Imagine 8.2 through 8.5 software. Subsequent statistical descriptions and analyses were performed on an Apple Powerbook computer using the 10.2.3 operating system. Statistical analysis was performed using Excel 2000 (Microsoft), Prism 3cx (Graphpad Software Inc.) and R open-source software programs (S programming language V 4.0, Bell Labs).

FOREST BIOMASS ESTIMATES AND EQUATIONS

Data estimating biomass in mature and secondary forest systems come from areas within each of the three transects. Mature forest estimates, by Laurance *et al.* (1999), are used because they have the largest spatial extent with 64 one ha plots surveyed for trees of >10 cm diameter breast height (DBH). Mature forest above-ground biomass is

reported as $356 \pm 47 \text{ Mg ha}^{-1}$, with data ranging from 492 to 231 $\text{Mg biomass ha}^{-1}$.

Below-ground tree biomass is not quantified in these estimates.

Secondary forest biomass in this study is estimated using a bilinear regression equation that changed slope, *i.e.*, rate of biomass sequestration, at 12 years of age. For secondary forests of two to 12 years of age sequestered carbon at the rate of $9.1 \pm 0.82 \text{ Mg biomass ha}^{-1} \text{ yr}^{-1}$ while secondary forests older than 12 years sequestered carbon at the rate of $5.9 \pm 0.53 \text{ Mg biomass ha}^{-1} \text{ yr}^{-1}$ (Steininger 2001). Below-ground tree biomass is not estimated nor is dead standing biomass. Together these factors may represent as much as 33% of the total biomass (Brown and Lugo 1990). Young secondary forests have a highly variable initial carbon sequestration rate associated with previous land tenure (Steininger 2001, Uhl *et al.* 1988, Mesquita 1995, Mesquita *et al.* 2001). Older secondary forest stands sequester carbon at decreased rates but have less variability in biomass regardless of their previous land tenure (Steininger 2001). This bilinear equation does not accumulate above-ground biomass until the second year of abandonment. Regrowth forest sequestration rates are compared to previous reports in Figure 3.2 (Uhl *et al.* 1988, Brown & Lugo 1990, Honzák *et al.* 1996). Typically, these studies measure above-ground biomass via allometric equations based on tree height, DBH and the specific gravity of the wood. Below-ground biomass has been estimated as being 19- 33% of the above-ground biomass present by other studies (Uhl *et al.* 1988, Brown & Lugo 1990, Honzák *et al.* 1996).

The proportioning of carbon emissions that are allocated to combustion, decomposition and absorption are collected from the Manaus region (Fearnside 1992,

Fearnside *et al.* 1993, Carvalho *et al.* 1995). Deforested mature forest biomass is partitioned with 27.4% being allocated to combustion, 70.0% allocated to decay and

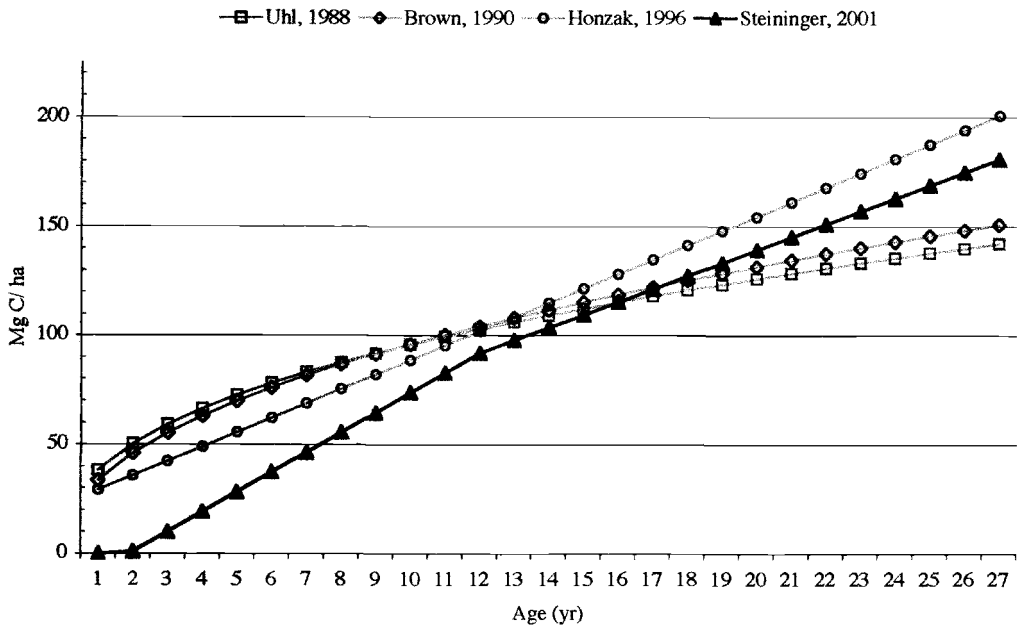


Figure 3.2 Comparison of different published biomass sequestration equations for tropical secondary forest systems. The equations of Uhl *et al.* (1988), Brown and Lugo (1992), and Honzák *et al.* (1996) have been altered by -25% to compensate for below-ground tree biomass not estimated in this study.

2.6% remains on the ground. Secondary forests are proportioned for combustion and decay with 60% and 40%, respectfully (Steininger *in review*).

TIME SERIES MODEL

The methodology for quantifying total above-ground biomass (TAGB) consists of three parts. The first part geo-rectifies MSS, TM and ETM+ satellite imagery within the same coordinate systems as ground data. The second part classifies satellite imagery into discrete classes for each annual satellite scene. The third part is an aging function that tracks individual pixels through time, recording the age of all non-mature forest pixels (*i.e.* 1977, 1985- 1999).

Georectification

Four differential global positioning system (dGPS) base stations were established in 2000 with two Magellan Promark X global positioning system (GPS) units. Base stations could record data continuously for 10 hours. One base station is located at the BDFFP central office (59°58' 45''W, 3°12'46''S) and one base station is at each of the three field sites, *Favienda Dimona*, *Porto Allegro* and *Collosa Favienda* (Figure 3.3). At this time, early 2000, the United States government offset all GPS satellites ($\pm 100\text{m}$). A system of post-process differential correction was required to obtain sub-pixel accuracy ($\pm 30\text{m}$) and produce accurate readings ($\pm 1\text{m}$). From January 17th until April 30th, 2000 over 23,000 differentially corrected GPS locations were collected at identifiable forest

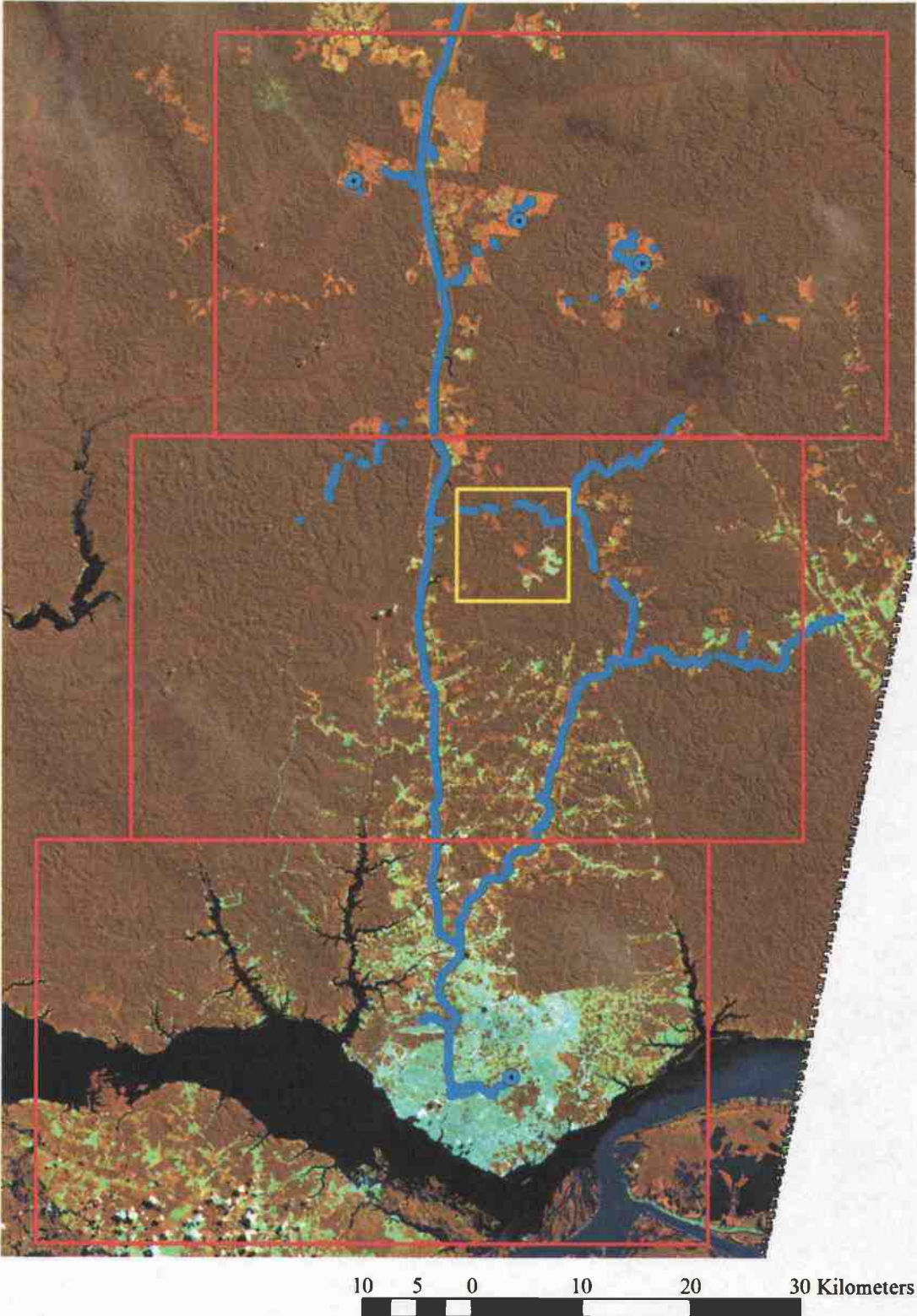


Figure 3.3 Spatial extent of ‘good’ differentially corrected global positioning system points. Blue circle with a black dot are established base station locations. Blue points are selected dGPS points collected.

fragment edges, road intersection, prominent water bodies and passable roads. GPS base stations and field units were equipped with multi-channel resistant antennas capable of being elevated to 15 m to increasing satellite signal clarity in heavy canopy habitats. Collected points were differentially corrected. These differentially corrected GPS points (dGPS) were filtered by their recorded positional accuracy as estimated from the strength of the signal from particular satellites and their spatial orientation in the horizon. At least four GPS satellites were needed to have an accurate GPS reading. 3,500 points were selected and transferred into a geographic information system (GIS) in UTM for zone 21 South.

On July 10th, 1999, Dana Slaymaker and NASA crew flew over the BDFFP field sites with aerial videography equipment. Approximately 85% of the BDFFP field sites were recorded with a minimum resolution of 2 m in a 800 m wide image path. Seven aerial videographic mosaics of the consecutive flight paths were combined via geo-positioning and spectral matching algorithms to produce a mosaic image of the BDFFP field sites (Hayward 2001). Corrected dGPS points were then used via a 2nd order polynomial transformation to further increase positional accuracy (Figure 3.4). This geo-referenced video-mosaic was then used in conjunction with corrected dGPS points to geo-rectify ETM+, TM and MSS satellite imagery. The Geo-referenced video-mosaic was also used to verify the results of the 1999 age map.

An August 9th 1999, ETM+ image was acquired via the Tropical Rain Forest Data Center (<http://www.bsrsi.msu.edu>). One of the improvements of ETM+ imagery is the spatial accuracy of an image is much more precise (± 50 m as compared to previous TM imagery ± 2 km). Spatial error was further reduced via 2nd order polynomial

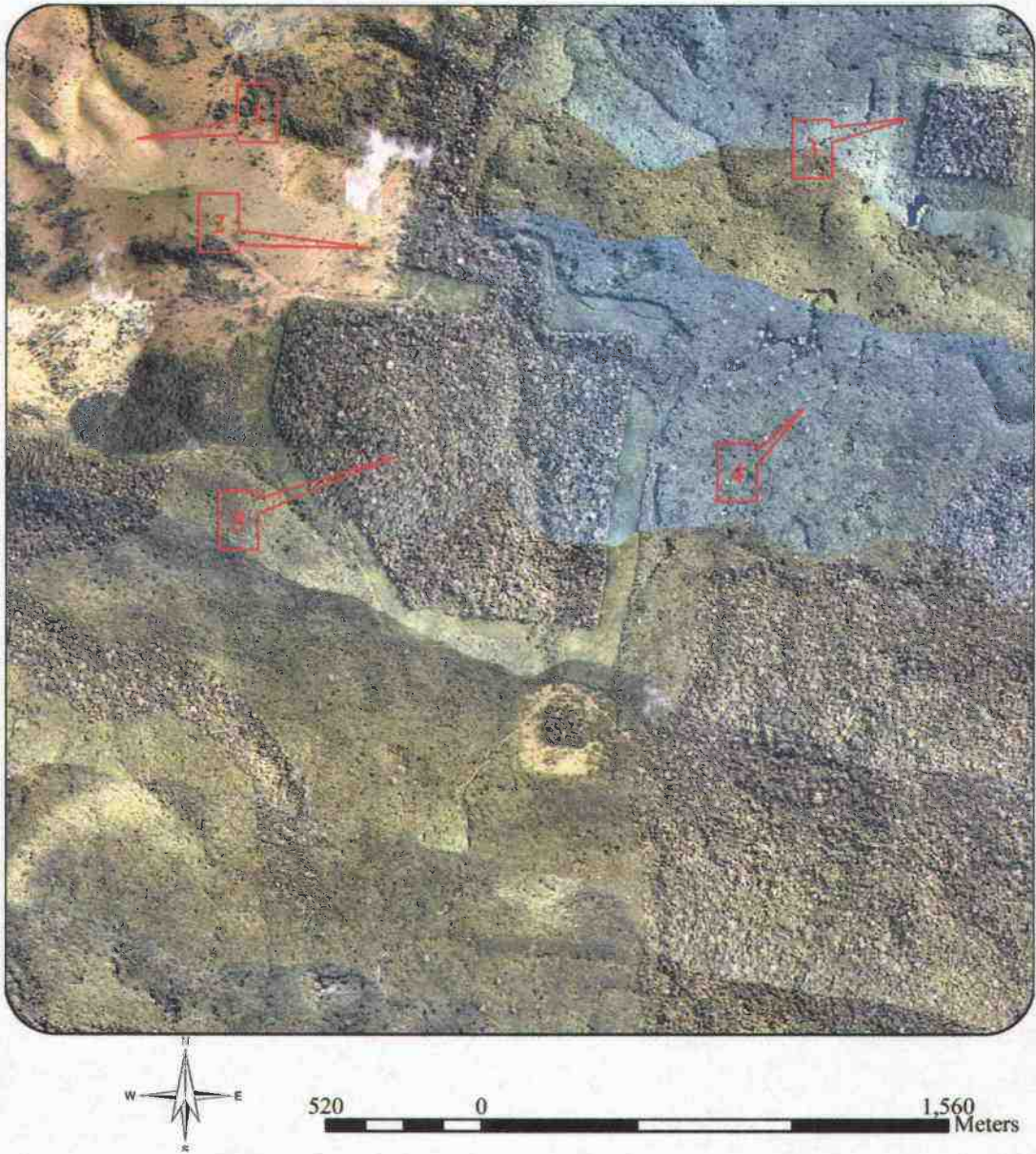


Figure 3.4 Geo-referenced aerial video mosaic used for the validation of Landsat ETM+ supervised classification. Early and late pasture (1 and 2), early and late secondary forest (3 and 4) and mature forest habitats (5) are identified and verified.

transformation with the aerial video-mosaic and corrected dGPS points. Prominent features such as roads, forest edges and water bodies are matched between the geo-referenced video-mosaic, selected dGPS points and the 1999 ETM+ satellite image. TM & MSS satellite imagery was geo-corrected to the 1999 ETM+ image or a satellite image that has been successfully geo-corrected of the closest date (*e.g.* 1997 was matched to 1999, 1996 was matched to the 1997 geo-corrected satellite image, etc.). Two of the TM images were for only the NE quadrant (1985 & 1990). The NE quadrant did not fully cover all of Transect 3. All images were visually inspected and corrected if spatial incongruities were found in the study area.

Classification

TM and ETM+ Imagery were classified in chronological order using a supervised maximum likelihood classification based on the composition of bands 3, 4 and 5 (red, near infrared and middle infrared). Imagery of a given year's satellite data is classified into the following classes: high forest, low forest, *baixo* forest, *igrape*, late stage regrowth, early stage regrowth, late pasture, early pasture/bare, white water river, black water river, cloud and cloud shadow. This classification was subsequently reclassified into pasture, water, early regrowth, late regrowth and mature forest classes. 1977 MSS imagery data was classified using a supervised maximum likelihood analysis for bands 2, 3 and 4.

Consistently the misclassification of regrowth and pasture would occur in mature forest along elevation ridgelines with low sun angle (most of the cloud free imagery was collected before 10:00 am). A manual masking procedure was used to correct for these

misclassifications. Manual corrections were further verified by annual classified and multi-band images to validate any particular deforestation event. This chronological feature allowed for correcting of deforestation events that were undetected due to cloud cover and cloud shadow classifications. This manual correction occurred on 12% of the area classified as anthropogenically altered. The spatial heterogeneity of disturbance events would be lost if more traditional area based classification filters were used. Predominantly, these corrections occurred along elevation ridgelines.

Age Modeling

Age modeling is based on a sequential annual accounting for every pixel within the study area. It depends on the ecological succession from pasture to late stage regrowth forest and the corresponding changes in the composition of spectral reflectance values over time. Figure 3.5 shows satellite imagery of the successional sequence as a pasture is created from mature forest (areas 1 and 4), pasture succeeds to early regrowth forest (2) and early regrowth succeeds to late stage regrowth forest (3). When an area of land is cleared, it has high signature values in the red and near infrared spectrum as a result of increased light reflected off the denuded area. As the area develops more canopy cover, the amplitude of red and near-infrared band readings decreases because increased canopy cover absorbs more light. As the canopy closes, it forms a flat homogeneous ceiling layer that maintains the same spectral characteristics over time, but the regrowth forest actually gains biomass as its spectrally flat canopy is raised to 20- 25 m in height (Lucas *et al.* 2002). This growth phase can last for 15- 30 years. The difference between the biomasses of early-phase closed-canopy regrowth forest and

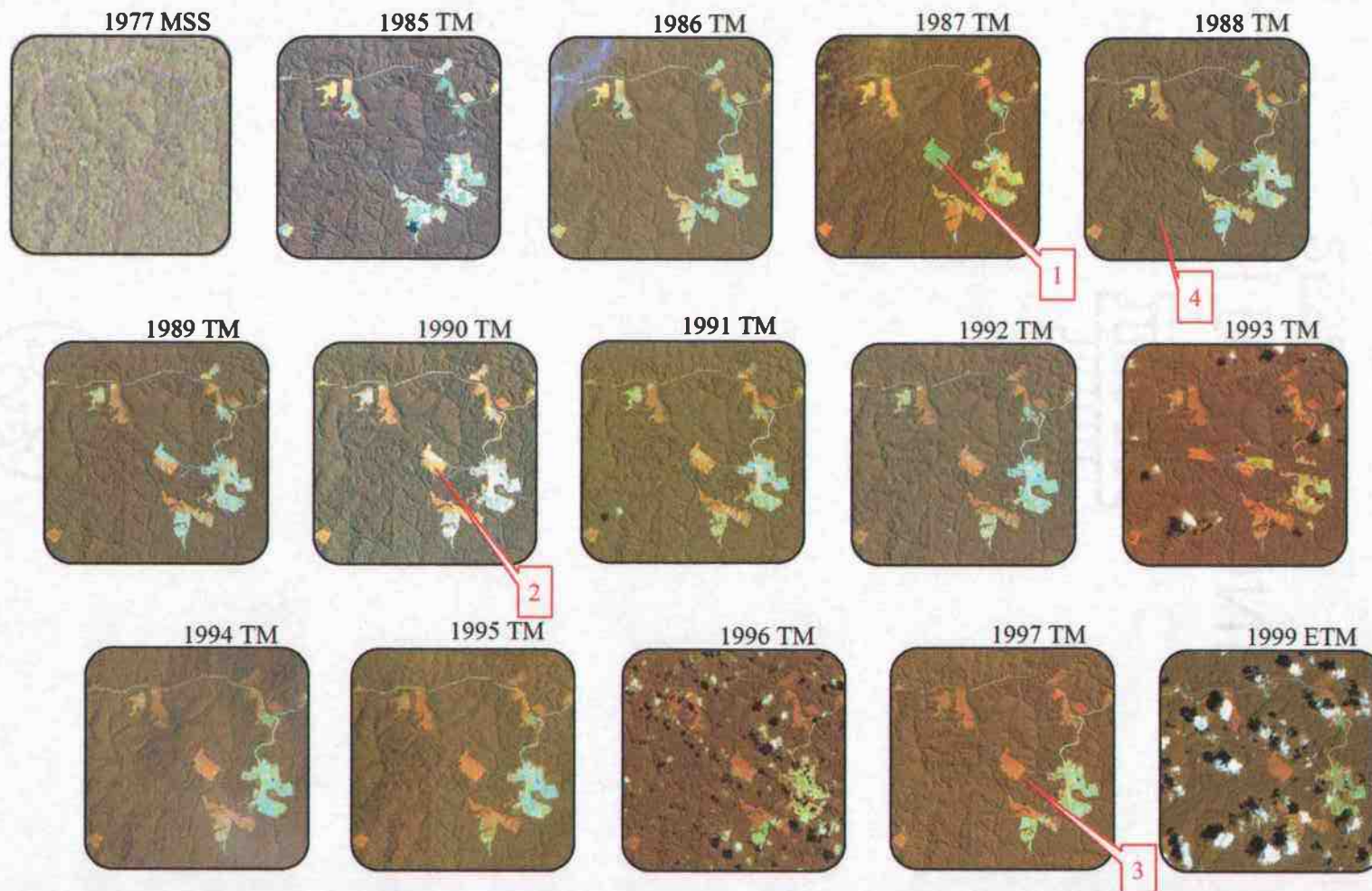


Figure 3.5 Chrono-sequence of Multi-Spectral Scanner, Thematic Mapper and Enhanced Thematic Mapper satellite imagery of a subsection of Transect 2 (MSS, TM and ETM, respectively). A boxed 1 is representative of spectra for pasture, 2 for early regrowth forest, 3 for late regrowth forest and 4 for mature forest (TM & ETM channels 4, 5, 3 = RGB, MSS channels 3, 4, 2 = RGB).

late-phase closed-canopy regrowth forest can exceed an order of magnitude, 50 or 150 Mg biomass ha⁻¹ (Figure 3.2). The difference in regrowth forest biomass explains why MSS, TM and ETM+ satellite imagery cannot accurately predict the standing above-ground biomass within these secondary forests.

Due to the pronounced spectral signature of pasture, it is possible to predict the age of a stand from chrono-sequenced annual imagery. The age model tracks the length of time since the last time a pasture spectra classification occurred. Time is a much better predictor of above-ground biomass relative to single year satellite imagery (Nepstad *et al.* 1991).

Classified 1977 MSS imagery and classified 1985 TM imagery were compared to produce an initial age map (Figure 3.6). Pixels classified as late regrowth in 1977 and as late regrowth in 1985 received an age of 12. Pixels classified as early regrowth in 1977 and late regrowth in 1985 received an age of 11. Pixels classified as pasture in 1977 and late regrowth in 1985 received an age of 8. Pixels classified as pasture in 1985 received an age of 0. Pixels classified as early regrowth in 1985 received an age of 3. Pixels classified as late regrowth forest in 1985 but were mature forest in 1977 received an age of 5.

Every annual age map is corrected with a high water mask (1991) for the seasonal fluctuation of the Amazon and adjacent rivers. Areas below the highest water level are excluded. The high-water mask creates a consistent land area across all years studied. If the age map were not corrected for high water, there would be an increase in land classified as pasture during low-water years. Bare, open sandy riverbanks would be classified as pasture because these areas would reflect an indistinguishable spectral

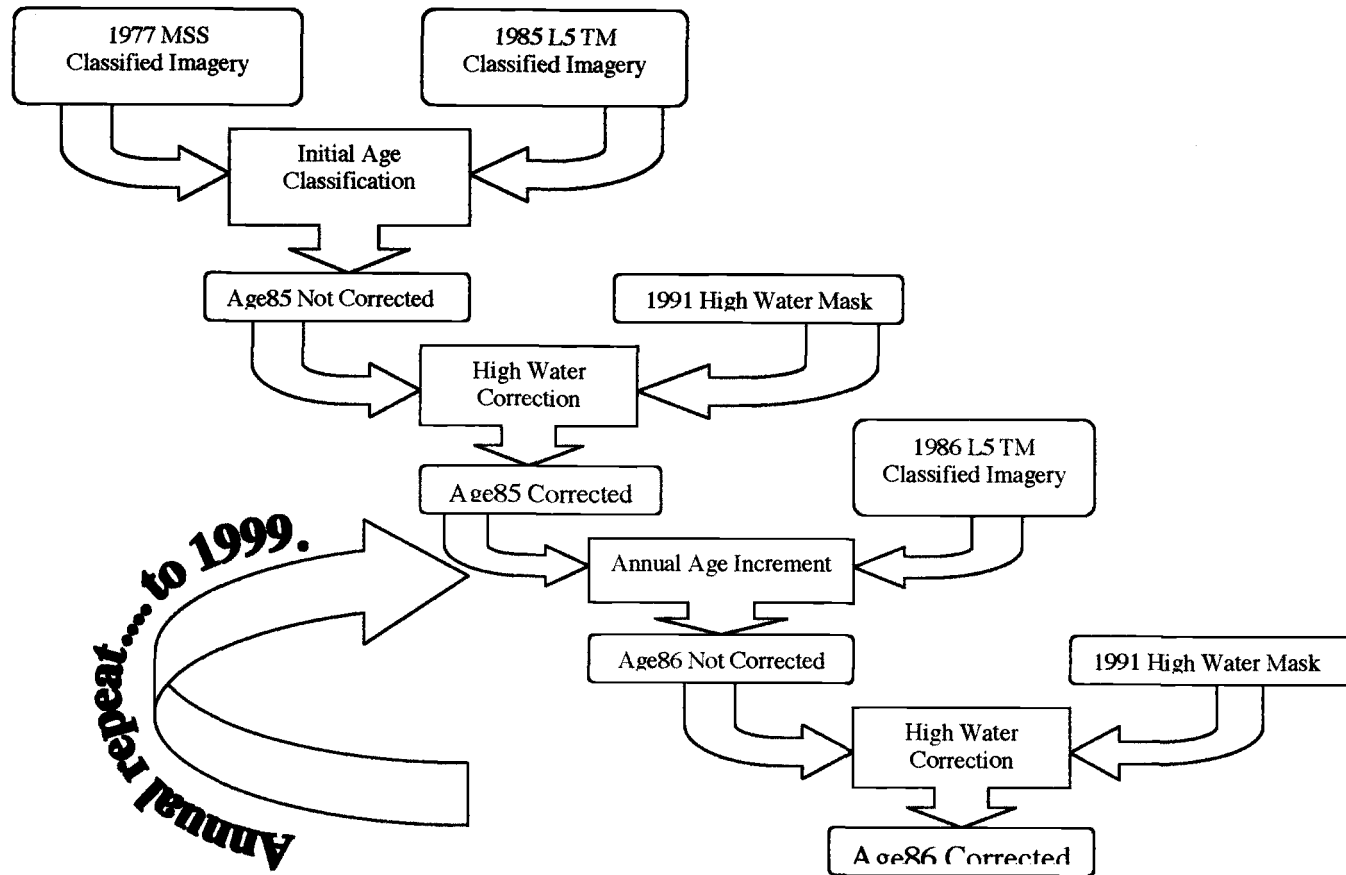


Figure 3.6 Flow chart of the sequential processes of the age mapping model. Age is initially estimated from 1977 MSS imagery and 1985 classified TM data. After age is determined, the next year's classified imagery is examined and then a new age map is produced for that year

signature in the red and near infrared. On average the waters of the port of Manaus fluctuate ± 8 m annually.

The high-water corrected age map for 1985 was then compared to 1986 classified imagery. If a pixel in the 1986 classified image was pasture, the age value received for that pixel is 0. If this cell was classified as any other class, the subsequent age received was the previous age + 1. This process produced the 1986 age map that is then corrected for high water. After being high-water mask corrected, the 1986 age map was compared to 1987's classified imagery to produce 1987's initial age map. The process of comparing high-water corrected age maps to the subsequent year's classified imagery to produce initial age maps that are subsequently corrected with the high-water mask continued until 1997 (Figure 3.6).

The sequence between the 1997's corrected age map and 1999's classified imagery is calculated for a two-year time step. Satellite imagery for 1998 was not available due to > 90% cloud cover on all available dates. The 1999 age map was corrected with the high-water mask. Age statistics were calculated for 1998 as the mean between age values for 1997 and 1999 results. Frequency distributions of age values were collected per annum from 1985 to 1999.

Figures 3.5, 3.7 and 3.8 explicitly show how the model recorded its findings across the different iterations of the satellite data for a smaller subsection of Transect 2 (Figure 3.1's yellow box). Specifically, Area 1 of Figure 3.5 is shown incrementally increasing its age from 1987 through 1999. Figure 3.7 displays the classified imagery in chronological sequence. Figure 3.8 displays the ages of regrowth forests, after high-water correction, through time in a spatially explicit fashion.

Analysis

The biomass of regrowth forests are calculated by multiplying a cell's age by the amount of biomass a typical regrowth forest of that age produces for a 30 x 30 m area.

Mature forest cells are calculated with mean mature forest biomass as reported by Laurance *et al.* 1999 (356 Mg biomass). Total above-ground biomass is the sum of mature forest and secondary forest biomasses. Results are reported for individual transects and all transects combined.

Linear regressions (least-mean-square) are used to characterize the behaviors of regrowth biomass, mature forest biomass, and total above-ground biomass through time (1985-1999). Transect 3's timeline is constrained to 1986-1999 due to the lack of complete satellite imagery. When statistics are derived for all transects combined, they also are presented for the years 1986 through 1999. To test whether accounting for age-specific regrowth forest above-ground biomass changed total above-ground biomass, a comparison of slopes test between total above-ground biomass and mature forest biomass is used (Zar 1984). If the slopes are not identical then it could be stated that age specific secondary forest biomass does affect the measurement of total above-ground biomass's rate of increase or decrease. If the slopes are found to be identical, the two regressions are then tested to see if they differed in elevation (comparison of two elevations, Zar 1984).

The average age of anthropogenically altered lands is quantified for all individual transects and for the total area studied. Deforestation rates are calculated and characterized via linear regression for each transect and for all transects combined.

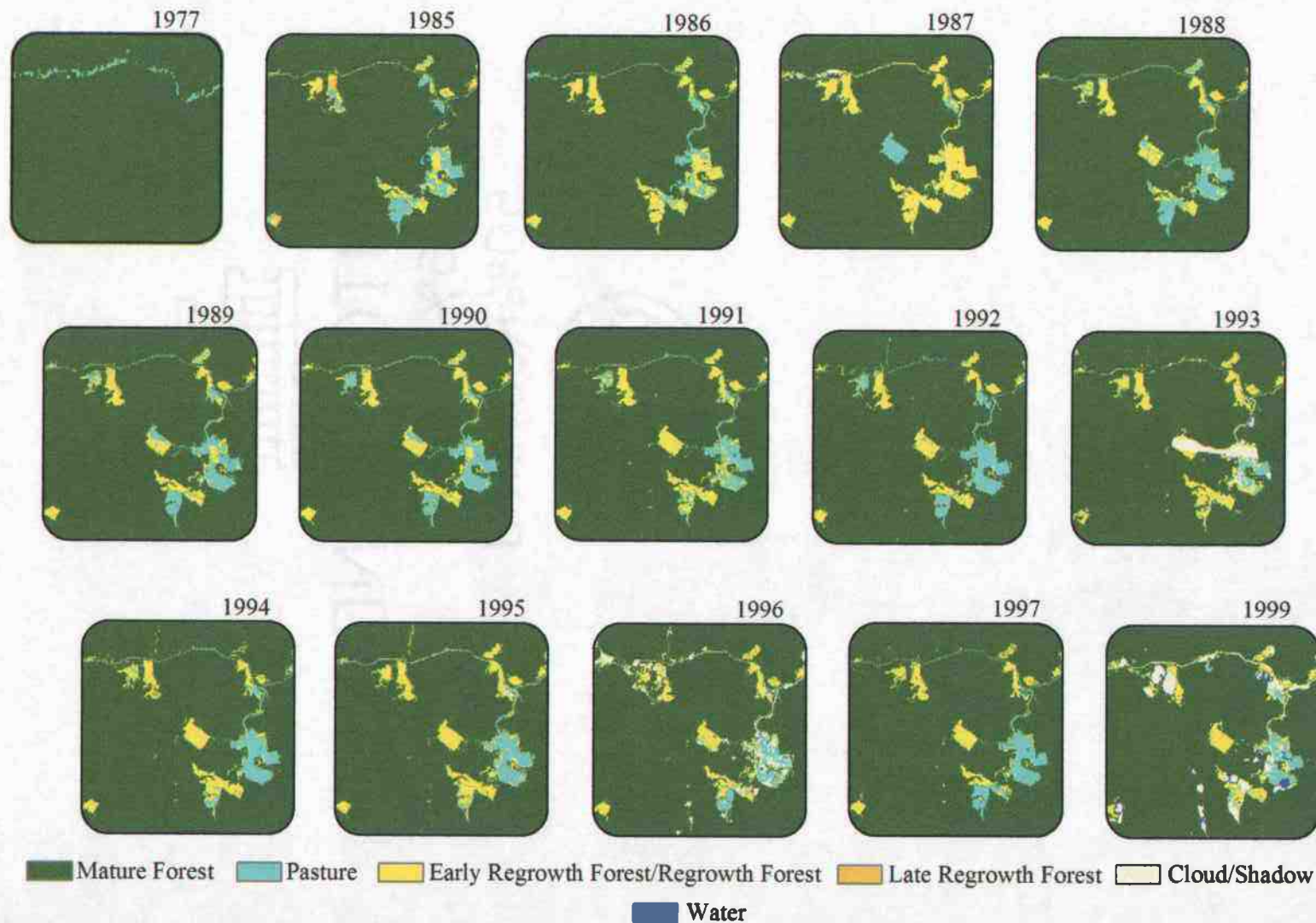


Figure 3.7 Classified and manual mask corrected chrono-sequence from a subsection of Transect 2.

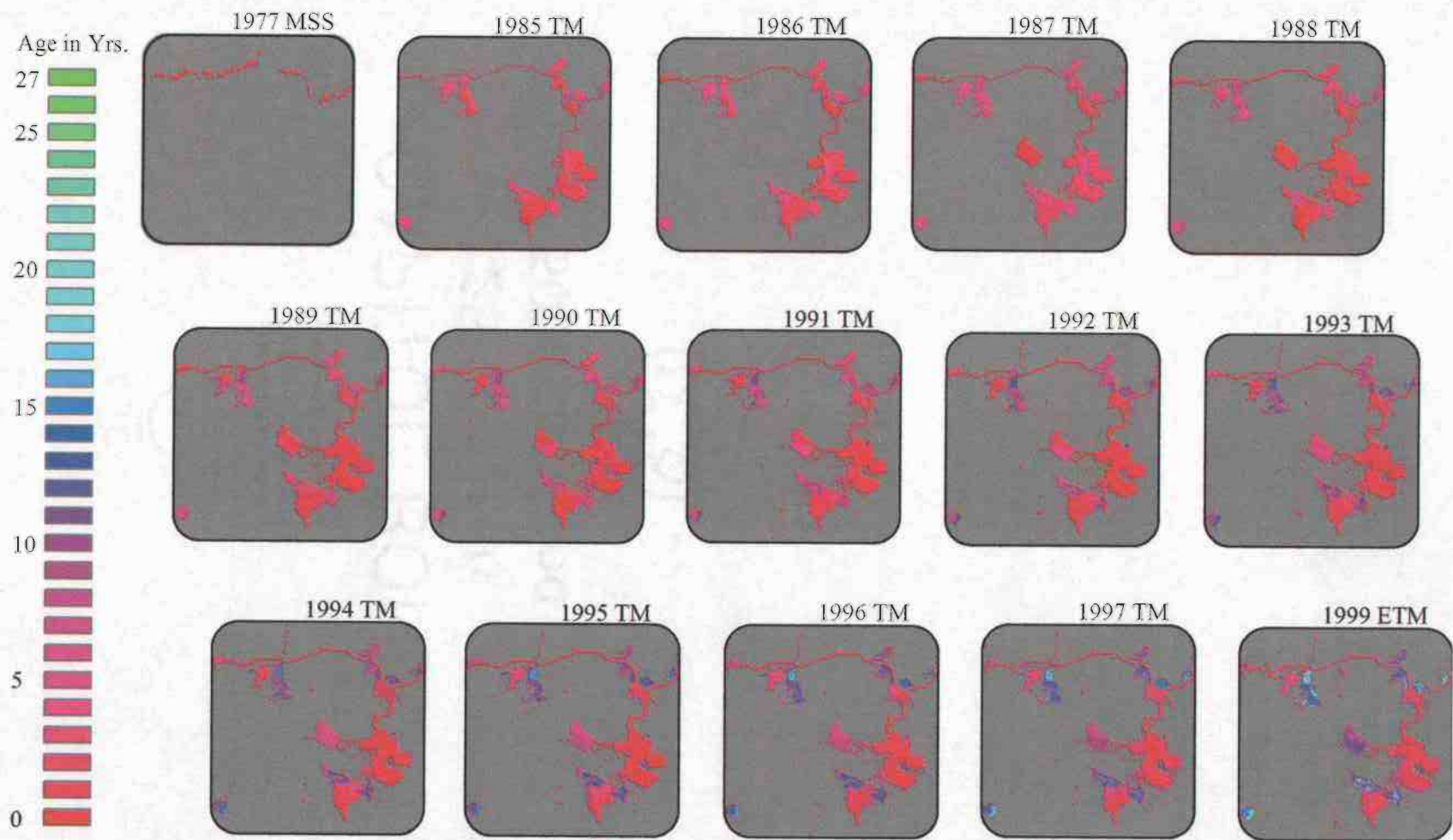


Figure 3.8 Age distribution map by year for the chrono-sequence of a subsection of Transect 2.

A change detection spreadsheet function accounted for the loss of particular ages of regrowth forest and all mature forest. The changes within an age class are then accounted for in terms of the biomass and area. Changes in mature forest are accounted for in biomass and area. The biomass amounts are used to estimate carbon emissions using the efficiencies derived for mature forests (Fearnside 1992) and regrowth forests (Steininger *in review*). Combustion, carry-over, and decay classes are also calculated. Combusted biomasses are contributed to the emissions of that year. Carry-over biomass is transferred to the subsequent year's regrowth forest class. Biomass allocated to decay is placed in a nine-year exponential decay function where 99.98% of the biomass is released to the atmosphere and is accounted for in subsequent years emissions statistics. Because of the half-life decay function, it required three years to load the model for approximate estimate of decay emissions (at three years approximately 87.5% of the total decaying biomass is accounted for). The total carbon emissions are calculated and characterized using linear regression on a per annum basis for 1989 to 1998. Total carbon emissions are characterized by simple linear regression.

Regrowth forest's above-ground biomass sequestered per annum is calculated and compared to total carbon emissions to examine if regrowth forest's sequestration compensates for total carbon emissions within dynamic land tenure systems. Annual total carbon emissions are comprised of four factors;

- 1) Regrowth combustion of the year
- 2) Mature forest combustion of the year

3) The sum of the previous nine years for deforested regrowth decay in annual

$$\text{time dependent increments } (E = e^{-0.63*(\text{Year}-1)} + e^{-0.63*(\text{Year}-2)} + e^{-0.63*(\text{Year}-\dots)} + e^{-0.63*(\text{Year}-9)})$$

4) The sum of for the previous nine years for mature forest that were deforested

$$\text{in annual time dependent increments } (E = e^{-0.63*(\text{Year}-1)} + e^{-0.63*(\text{Year}-2)} + e^{-0.63*(\text{Year}-\dots)} + e^{-0.63*(\text{Year}-9)})$$

The annual sequestration of regrowth forests is compared to the annual total carbon emissions for 1989 to 1998. The year 1989 was chosen to allow for loading of the carbon emissions from decay. The analysis was concluded in 1998 because of the methodological constraint of not being able to estimate a given year's emissions without having the classified imagery of the subsequent year, *i.e.*, we would need satellite imagery from 2000 to quantify the changes that occurred in 1999.

RESULTS

The age model produced age estimates for anthropogenically altered lands (Figures 3.9 and 3.10). Using the age to calibrate above-ground biomass total, mature forest and regrowth forest above-ground biomass are presented for each transect and for the combined area of all three transects (Figure 3.11). The average ages are presented for each of transects and for the combined area of all three transects (Figure 3.12). Carbon emission from above-ground biomass ignition and decomposition are quantified through time (Figure 3.13).

DESCRIPTION OF TOTAL, MATURE FOREST AND REGROWTH FOREST ABOVE-GROUND BIOMASS

Transect 1. Total Above-ground Biomass is quantified as significantly increasing at the rate of $80,946 \text{ Mg yr}^{-1}$ with a standard deviation of $\pm 10,055 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.9847$, Table 3.1). Mature forest biomass significantly decreased at the rate of $44,262 \pm 11,966 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.9316$). Regrowth forest above-ground biomass significantly increased at a rate of $125,060 \pm 9,317 \text{ Mg yr}^{-1}$ ($p < 0.0001$, $R^2_{adj} = 0.9945$).

Transect 2. Total Above-ground Biomass is quantified as significantly decreasing at the rate of $279,399 \text{ Mg yr}^{-1}$ with a standard deviation of $\pm 36,088 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.9836$, Table 3.1). Mature forest above-ground biomass significantly decreased at the rate of $339,794 \pm 36,290 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.9887$). Regrowth forest above-ground biomass significantly increased at the rate of $60,278 \pm 9,960 \text{ Mg}$ ($p < 0.0001$, $R^2_{adj} = 0.9734$).

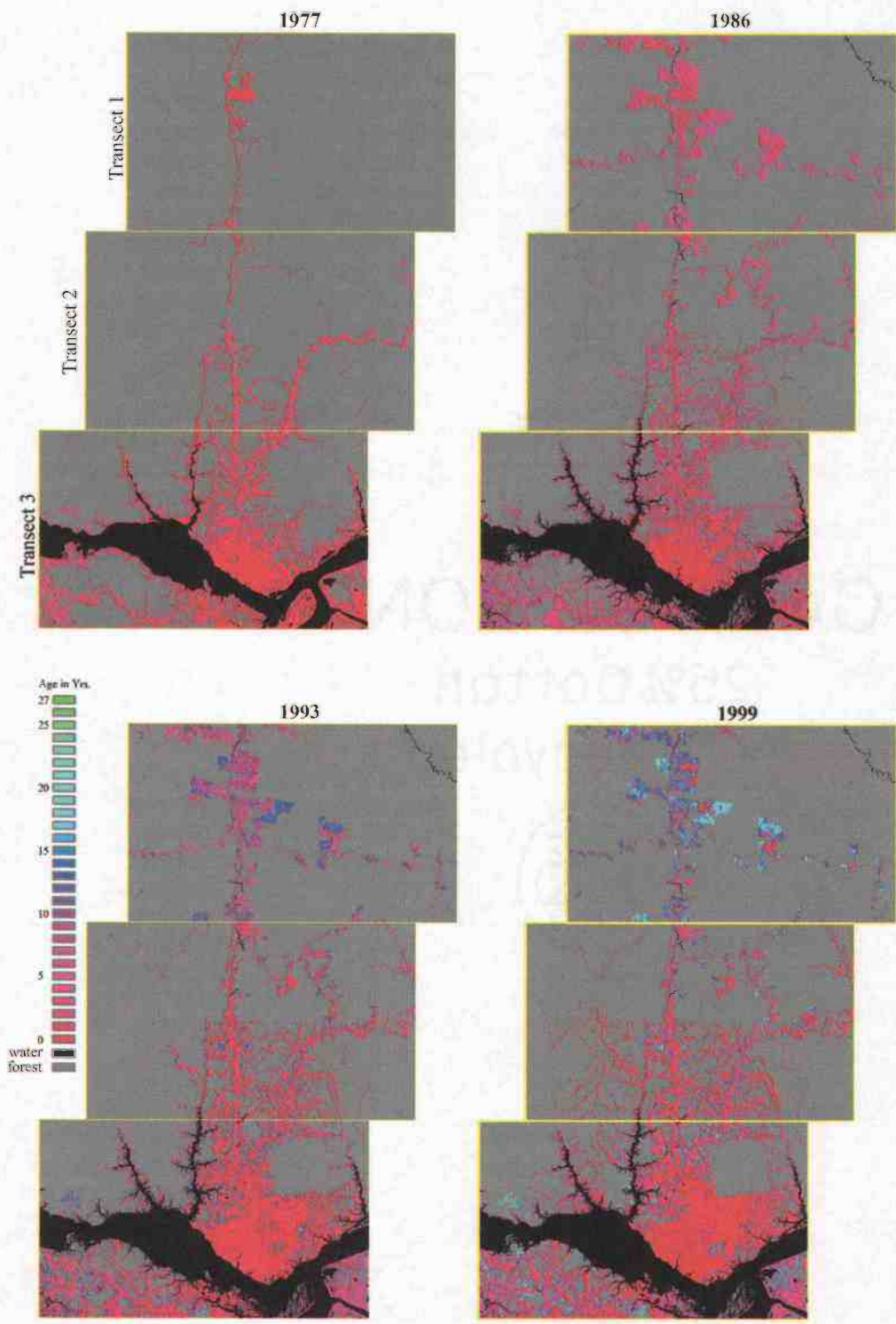


Figure 3.9 Results of the age based model for Transect 1, 2, and 3. 1977 MSS imagery was used to establish a base from which annual variability could be tracked on a pixel by pixel basis (1985 – 1999).

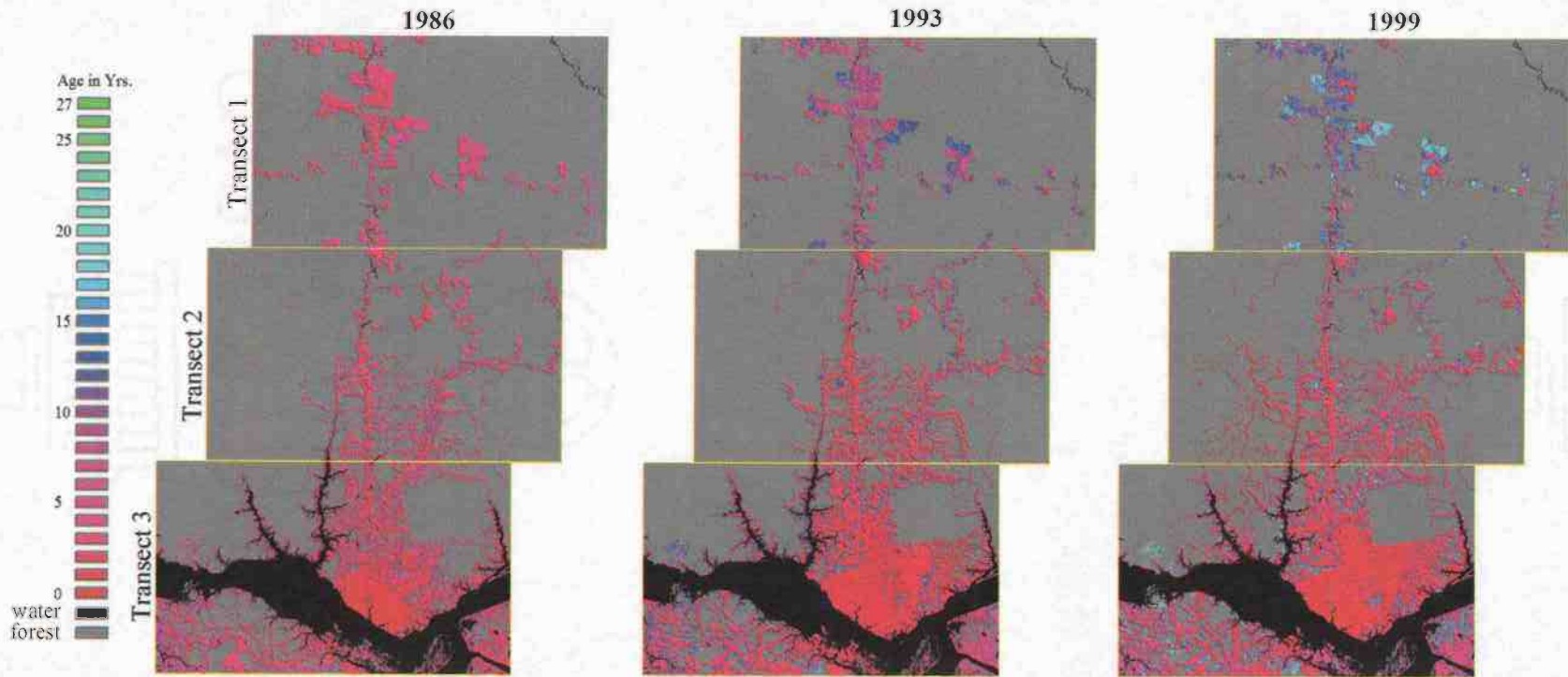


Figure 3.10 Results from the spatially explicit age model for the years 1986, 1993 and 1999. The years 1985 through 1999 were modeled on an annual increment tracking each cell's age through time.

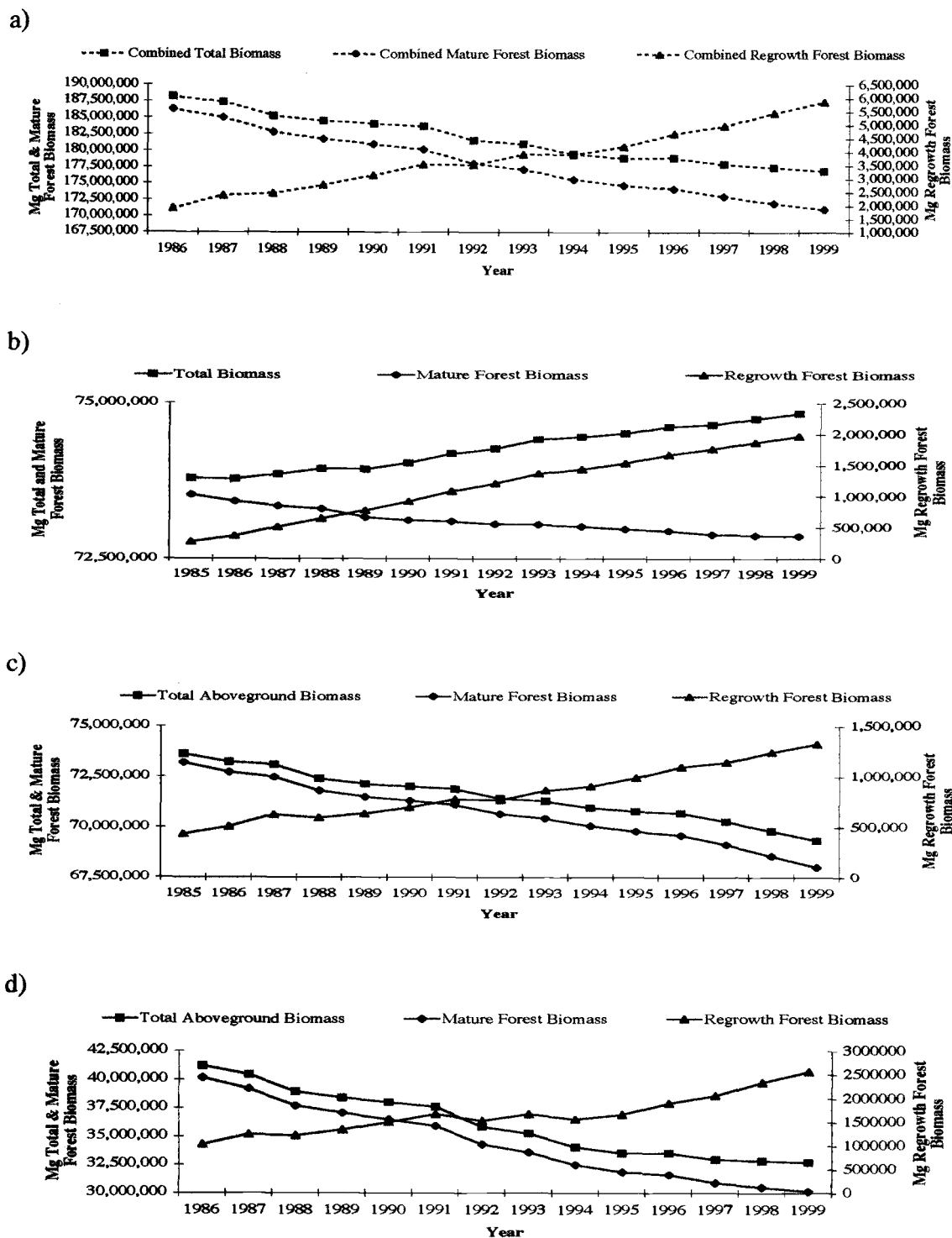


Figure 3.11 Above-ground biomass from 1986 to 1999. Total above-ground biomass (squares) and mature forest above-ground biomass (diamonds) reference the left y axis and regrowth above-ground forest biomass (triangles) reference the right y axis for combined transects (a), Transect 1 (b), Transect 2 (c) and Transect 3 (d).

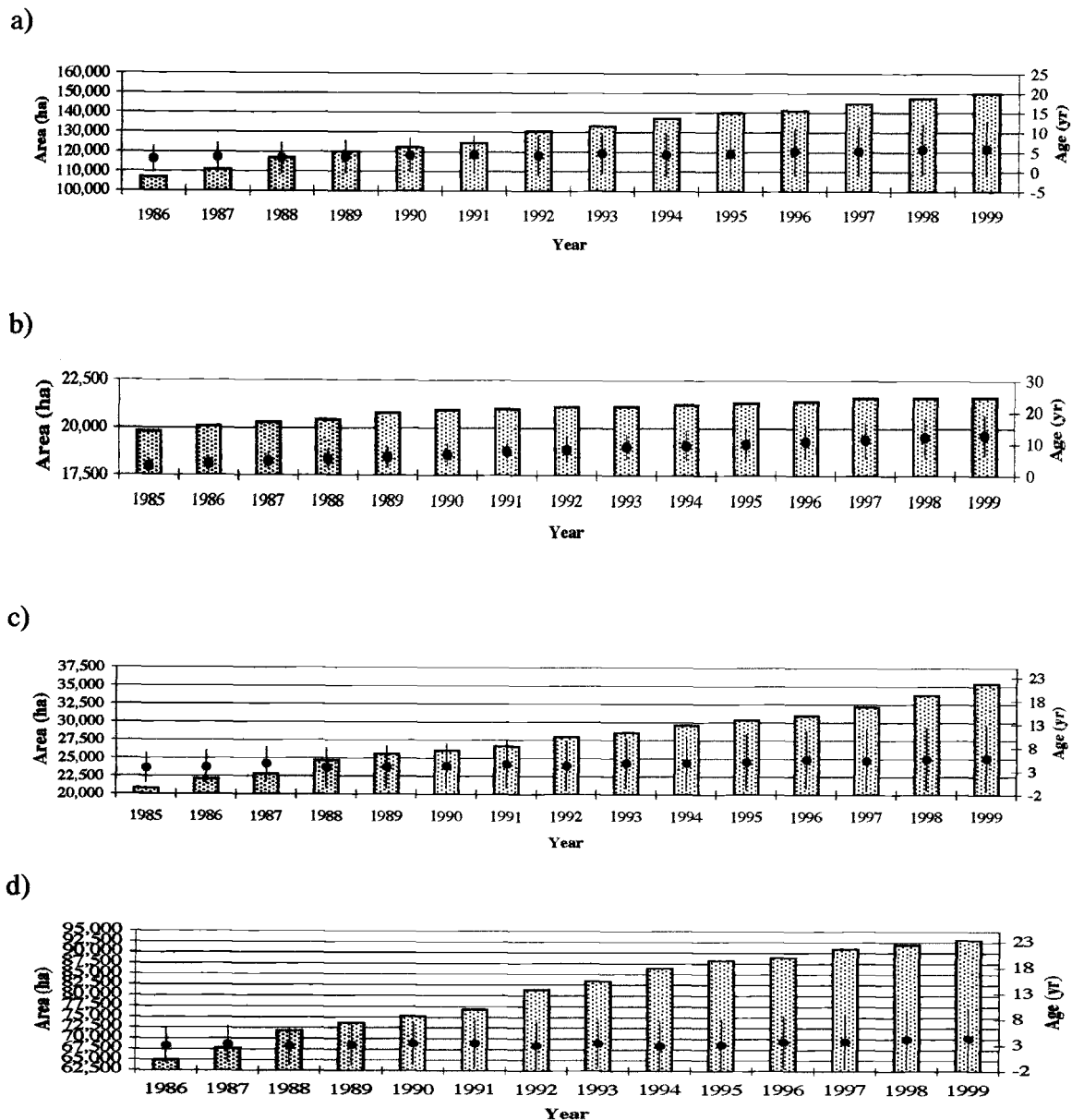


Figure 3.12 Average age with standard deviations and deforested areas. Average age per annum (filled circle) with \pm standard deviation (line) within the total deforested area (stippled box) for combined transects (a), Transect 1 (b), Transect 2 (c), and Transect 3 (d). Average age reference the right y axis. Deforested area references the left y axis.

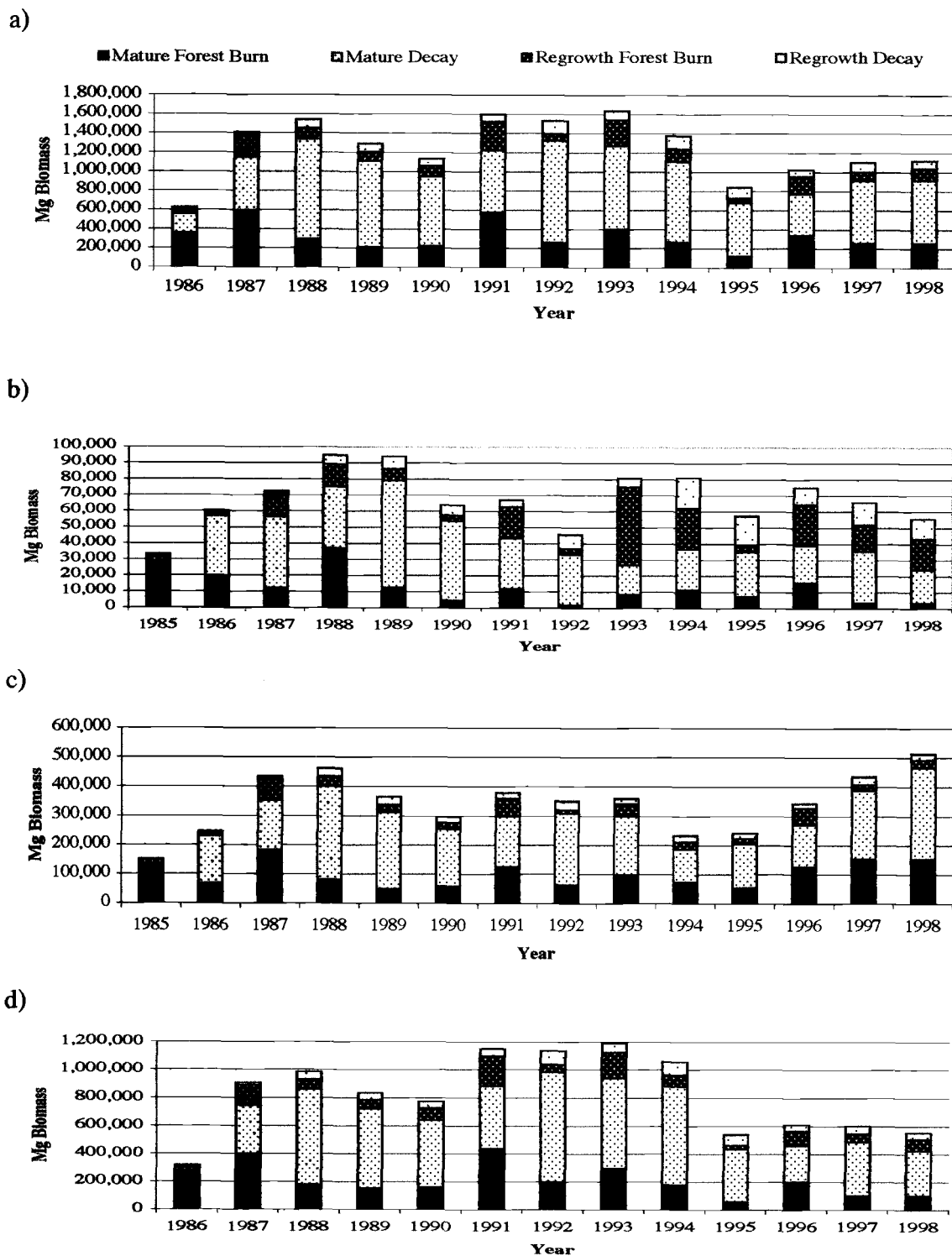


Figure 3.13 Biomass emissions in terms of ignition and decomposition in mature and regrowth forest types per annum for combined transects (a), Transect 1 (b), Transect 2 (c), Transect 3 (d).

Table 3.1 Linear Regression results for above-ground biomass (Mg) for combined transects, Transect 1, 2 and 3.

Location Type of Biomass	n	Coefficient	Standard Error	P-value	R²_{adj.}
Combined Transects					
Total Above-ground Biomass	14	-883,980	42,080	7.85E-11	0.9713
Mature Forest Above-ground Biomass	14	-1,166,185	33,230	1.83E-13	0.9895
Regrowth Forest Above-ground Biomass	14	282,205	11,158	8.87E-12	0.9801
Transect 1					
Total Above-ground Biomass	15	80,946	2,687	2.06E-13	0.9848
Mature Forest Above-ground Biomass	15	-44,262	3,197	3.70E-9	0.9316
Regrowth Forest Above-ground Biomass	15	125,060	2,490	2.83E-16	0.9945
Transect 2					
Total Above-ground Biomass	15	-279,399	9,645	3.40E-13	0.9836
Mature Forest Above-ground Biomass	15	-339,794	9,699	2.95E-14	0.9887
Regrowth Forest Above-ground Biomass	15	60,278	2,662	7.89E-12	0.9733
Transect 3					
Total Above-ground Biomass	14	-690,971	42,122	1.39E-9	0.9538
Mature Forest Above-ground Biomass	14	-789,704	33,169	1.81E-11	0.9775
Regrowth Forest Above-ground Biomass	14	97,001	10,102	5.54E-7	0.8752

Transect 3. Total above-ground biomass is quantified as significantly decreasing at the rate of $690,971 \text{ Mg yr}^{-1}$ with a standard deviation of $\pm 151,873 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.9538$, Table 3.1). Mature forest above-ground biomass significantly decreased at the rate of $789,704 \pm 119,593 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.9775$). Regrowth forest above-ground biomass significantly increased at the rate of $97001 \pm 36,423 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.8752$).

Combined. Total above-ground biomass is quantified as significantly decreasing at the rate of $-883,980 \text{ Mg yr}^{-1}$ with a standard deviation of $\pm 151,722 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.9713$, Table 3.1). Mature forest above-ground biomass decreased at the rate of $1,166,185 \pm 119,816 \text{ Mg}$ per annum ($p < 0.0001$, $R^2_{adj} = 0.9895$). Regrowth forest above-ground biomass increased at the rate of $282,204 \pm 40,231 \text{ Mg}$ ($p < 0.0001$, $R^2_{adj} = 0.8752$).

COMPARISON TO OF THIS MODEL TO MORE TRADITIONAL FOREST/ NON-FOREST METHODOLOGY

Transect 1. A comparison of the slopes of total above-ground biomass *versus* mature forest biomass shows that the slopes are different and highly significant ($F = 898.662$, $DF_n = 1$ $DF_d = 26$, $p < 0.0001$), meaning there is less than a 0.01% chance of randomly choosing data points with slopes this different.

Transect 2. A comparison of slopes between total above-ground biomass *versus* mature forest biomass is highly significant ($F = 19.497$, $DF_n = 1$ $DF_d = 26$, $p < 0.0002$), meaning there is 0.016% chance of randomly choosing points with slopes this different.

Transect 3. A comparison of slopes between total above-ground biomass *versus* mature forest biomass did not find the slopes to be significantly different ($F = 3.3914$, DF_n

= 1 DFd = 24, $p = 0.0780$) meaning there is 7.8% chance of randomly chosen data points with slopes this different. A comparison of the elevations of total above-ground biomass regression versus mature forest above-ground biomass regression found that the elevations are significantly different ($F = 53.0359$, $DFn = 1$ DFd = 25, $p < 0.0001$) meaning that there is less than 0.01% chance of randomly choosing data points with elevations this different.

Combined. A comparison of the slopes of total above-ground biomass *versus* mature forest biomass shows that the slopes are different and highly significant ($F: 27.7009$, $DFn = 1$ DFd = 24, $p < 0.0001$), meaning there is less than a 0.01% chance of randomly choosing data points with slopes this different.

CHARACTERIZATION OF TENURE SYSTEMS

Transect 1. The mean annual age of regrowth forest tenure is increasing significantly at the rate of 8.5 months with a standard deviation of 11 days per annum for all anthropogenically altered areas ($p < 0.0001$, $R^2_{adj} = 0.9976$, Table 3.2). The average age increased consistently throughout the length of the study. The total area of regrowth forest systems is also increasing significantly at the rate of 124 ha yr⁻¹ with a standard deviation of ± 34 ha per annum ($p < 0.0001$, $R^2_{adj} = 0.9316$, Table 3.3).

Transect 2. The mean annual age of regrowth forest systems is increasing significantly at the rate of 1.8 months yr⁻¹, with a standard deviation of ± 23 days per annum for all anthropogenically modifies areas ($p < 0.0001$, $R^2_{adj} = 0.8511$, Table 3.2). The average age shifted between increasing and decreasing throughout the study. The total area

Table 3.2 Results of the linear regression for time *versus* average regrowth forest age (years) for combined transects, Transect 1, Transect 2 and Transect 3.

Location	n	Coeffecient	Standard Error	P-value	$R^2_{adj.}$
Combined transects	14	0.1959	0.0160	3.7E-8	0.9201
Transect 1	15	0.7132	0.0094	1.43E-18	0.9975
Transect 2	15	0.1536	0.0171	5.99E-7	0.8511
Transect 3	14	0.0955	0.0216	0.0008	0.5883

Table 3.3 Results for the linear regression of time *versus* area deforested (ha) for combined transects, Transect 1, Transect 2 and Transect 3.

Location	n	Coeffecient	Standard Error	P-value	$R^2_{adj.}$
Combined transects	14	3,276	93	1.83E-13	0.9895
Transect 1	15	124	9	3.70E-9	0.9316
Transect 2	15	954	27	2.96E-14	0.9887
Transect 3	14	2,218	93	1.81E-11	0.9775

classified as anthropogenically altered is increasing significantly at the rate of $954 \text{ ha yr}^{-1} \pm 102 \text{ ha per annum}$ ($p < 0.0001$, $R^2_{adj} = 0.9887$, Table 3.3).

Transect 3. The mean annual age of regrowth forest systems is increasing at the rate of 3.5 days yr^{-1} , with a standard deviation of $\pm 28 \text{ days per annum}$ for all anthropogenically modified systems ($p = 0.0008$, $R^2_{adj} = 0.5883$, Table 3.2). The average age meandered between increasing and decreasing throughout the study. The total area of anthropogenically modified areas increased at the rate of $2,218 \text{ ha yr}^{-1}$, with a standard deviation of $\pm 349 \text{ ha per annum}$ ($p < 0.0001$, $R^2_{adj} = 0.9775$, Table 3.3).

Combined. The mean annual age increased at the rate of $2.35 \text{ months yr}^{-1}$, with a standard deviation of $\pm 20.7 \text{ days per annum}$ for all the anthropogenically altered areas ($p < 0.0001$, $R^2_{adj} = 0.9201$, Table 3.2). The average age increased and decreased throughout the length of the study. The total area of anthropogenically modified areas increased at the rate of $3,276 \text{ ha yr}^{-1}$, with a standard deviation of $\pm 337 \text{ ha. per annum}$ ($p < 0.0001$, $R^2_{adj} = 0.9895$, Table 3.3).

CHARACTERIZATION OF CARBON EMISSIONS

Transect 1. The total carbon emissions are found to be decreasing insignificantly at the rate of $736 \text{ Mg carbon (C)}$ with an annual standard deviation of $\pm 2,253 \text{ Mg C}$ (Table 3.4). The mature forest's carbon emissions accounted for $624,485 \text{ Mg biomass}$. Mature forests represented a total of 65.9% of the carbon emissions, with 19.1% from ignition and 46.8% from decomposition (Table 3.5). Carbon emissions of regrowth forest accounted for $323,862 \text{ Mg biomass}$ and represented 34.1% of the total carbon emissions, with 22.1% coming from combustion and 12% from decay (Table 3.5).

Table 3.4 Results of Linear regression for time *versus* carbon emissions (Mg) for combined transects, Transect 1, Transect 2 and Transect 3.

Location	n	Coeffecient	Standard Error	P-value	R²_{adj.}
Combined transects	11	--19,838	12,851	0.1612	0.1332
Transect 1	11	-736	751	0.3561	0.0046
Transect 2	11	1,220	4,083	0.7720	0.1002
Transect 3	11	-20,939	10,048	0.0668	0.2505

Table 3.5 Carbon emissions (Mg biomass) by source for 1986 to 1998 for Transect 1, Transect 2, Transect 3 and combined transects.

Combined transects

	1986 %	1987 %	1988 %	1989 %	1990 %	1991 %	1992 %	1993 %	1994 %	1995 %	1996 %	1997 %	1998 %	Total	%
Mature Forest Burn	356,198 67.1	598,567 42.4	296,299 19.2	217,595 16.8	223,626 19.7	577,471 36.2	263,588 17.2	404,218 24.7	266,705 19.4	121,571 14.6	343,134 33.3	260,064 23.5	260,064 23.1	4,189,100	25.8
Mature Forest Decay	201,830 32.3	555,912 39.4	1,042,549 67.6	899,758 69.6	727,829 64.1	649,569 40.7	1,062,430 69.3	867,915 53.1	843,070 61.3	557,819 66.3	433,312 42.0	653,473 59.2	658,195 58.6	9,153,662	56.3
Regrowth Forest Burn	57,233 9.2	234,209 16.6	114,237 7.4	92,983 7.2	112,061 9.9	296,384 18.6	70,599 4.6	270,055 16.5	130,066 9.5	50,456 6.0	181,899 17.6	94,108 8.5	128,193 11.4	1,832,486	11.3
Regrowth Forest Decay	8,819 1.4	23,487 1.7	89,813 5.8	82,966 6.4	72,487 6.4	73,597 4.6	135,594 8.8	91,330 5.6	135,683 9.9	111,180 13.2	72,371 7.0	96,686 8.8	79,628 7.1	1,073,642	6.6
Total	624,080	1,412,175	1,542,898	1,293,322	1,136,004	1,597,022	1,532,211	1,633,518	1,375,524	841,026	1,030,717	1,104,311	1,126,080	16,248,890	

Transect 1

	1985 %	1986 %	1987 %	1988 %	1989 %	1990 %	1991 %	1992 %	1993 %	1994 %	1995 %	1996 %	1997 %	1998 %	Total	%
Mature Forest Burn	28,971 86.6	19,998 33.1	12,413 17.1	37,205 39.2	12,703 13.5	4,907 7.7	12,220 18.2	2,195 4.8	8,401 10.4	11,509 14.2	7,488 13.1	15,951 21.2	3,547 5.4	3,547 6.3	181,057	19.1
Mature Forest Decay	0 0.0	37,006 61.2	44,049 60.8	37,881 39.9	66,465 70.6	49,459 77.4	30,998 46.1	31,109 68.2	18,358 22.7	25,324 31.2	27,291 47.6	23,161 30.9	31,925 48.5	20,400 36.5	443,428	46.8
Regrowth Forest Burn	4,471 13.4	2,005 3.3	14,622 20.2	14,221 15.0	7,408 7.9	3,271 5.1	19,811 29.5	3,617 7.9	48,435 60.0	25,531 31.4	4,619 8.1	25,432 33.9	16,637 25.3	19,510 34.9	209,590	22.1
Regrowth Forest Decay	0 0.0	1,490 2.5	1,413 1.9	5,581 5.9	7,531 8.0	6,235 9.8	4,208 6.3	8,707 19.1	5,560 6.9	18,925 23.3	17,970 31.3	10,523 14.0	13,729 20.9	12,401 22.2	114,272	12.0
Total Emiss. Mg BM	33,442	60,500	72,497	94,868	94,107	63,873	67,237	45,629	80,754	81,288	57,368	75,067	65,839	55,859	948,347	

Transect 2

	1985 %	1986 %	1987 %	1988 %	1989 %	1990 %	1991 %	1992 %	1993 %	1994 %	1995 %	1996 %	1997 %	1998 %	Total	%
Mature Forest Burn	129,033 85.4	67,765 27.5	183,796 42.3	82,127 17.8	49,680 13.6	59,565 20.0	125,258 33.1	62,691 17.9	98,342 27.4	74,182 31.7	54,219 22.3	126,821 37.0	153,575 35.3	153,575 29.9	1,420,629	29.5
Mature Forest Decay	0 0.0	164,823 66.8	168,973 38.9	319,263 69.0	264,538 72.6	195,729 65.7	173,952 46.0	246,977 70.5	203,568 56.6	114,765 49.0	151,819 62.6	144,998 42.3	234,038 53.8	312,986 60.9	2,696,428	56.0
Regrowth Forest Burn	21,986 14.6	6,755 2.7	75,757 17.4	32,891 7.1	25,330 6.9	21,604 7.3	61,615 16.3	11,411 3.3	38,942 10.8	22,879 9.8	17,841 7.4	55,762 16.3	20,908 4.8	26,930 5.2	440,611	9.2
Regrowth Forest Decay	0 0.0	7,329 3.0	5,916 1.4	28,210 6.1	25,069 6.9	20,978 7.0	17,690 4.7	29,383 8.4	18,495 5.1	22,228 9.5	18,726 7.7	15,306 4.5	26,191 6.0	20,043 3.9	255,565	5.3
Total	151,019	246,672	434,442	462,491	364,617	297,876	378,515	350,462	359,347	234,055	242,605	342,886	434,711	513,534	4,813,233	

Transect 3

	1986 %	1987 %	1988 %	1989 %	1990 %	1991 %	1992 %	1993 %	1994 %	1995 %	1996 %	1997 %	1998 %	Total	%
Mature Forest Burn	268,434 84.7	402,357 44.4	176,966 18.0	155,212 18.6	159,154 20.6	439,993 38.2	198,703 17.5	297,475 24.9	181,013 17.1	59,864 11.1	200,362 32.7	102,942 17.1	102,942 18.5	2,745,418	25.7
Mature Forest Decay	0 0.0	342,890 37.9	685,405 69.5	568,754 68.1	482,641 62.3	444,619 38.6	784,344 69.0	645,990 54.1	702,981 66.3	378,709 70.0	265,153 43.3	387,510 64.2	324,809 58.3	6,013,806	56.4
Regrowth Forest Burn	48,473 15.3	143,830 15.9	67,126 6.8	60,245 7.2	87,186 11.3	214,959 18.7	55,570 4.9	182,678 15.3	81,657 7.7	27,996 5.2	100,706 16.4	56,563 9.4	81,754 14.7	1,208,742	11.3
Regrowth Forest Decay	0 0.0	16,158 1.8	56,022 5.7	50,386 6.0	45,275 5.8	51,699 4.5	97,503 8.6	67,275 5.6	94,530 8.9	74,484 13.8	46,542 7.6	56,746 9.4	47,184 8.5	703,804	6.6
Total	316,908	905,236	985,520	834,598	774,255	1,151,270	1,136,120	1,193,417	1,060,181	541,053	612,764	603,761	556,688	10,671,771	

Transect 2. Total carbon emissions are found to be insignificantly increasing at the rate of 1220 Mg C, with an annual standard deviation of $\pm 12,912$ Mg C (Table 3.4). Carbon emissions of mature forests accounted for 4,117,057 Mg biomass and represented a total of 85.5% of the total carbon emissions, with 29.5% from ignition and 56% from decomposition of the total carbon emissions (Table 3.5). Carbon emissions from regrowth forests accounted for 696,176 Mg biomass and represented 14.5% of the total carbon emissions, with 9.2% from ignition and 5.3% from decomposition.

Transect 3. Total carbon emissions are found to be insignificantly decreasing at the rate of 20,939 Mg C, with an annual standard deviation of $\pm 31,775$ Mg C (Table 3.4). Carbon emissions from mature forests accounted for 8,759,224 Mg biomass and represented a total of 82.1% of the total carbon emissions, with 25.7% from ignition and 56.4% from decomposition (Table 3.5). Carbon emissions from regrowth forest accounted for 1,912,546 Mg biomass and represented 17.9% of the total carbon emissions, with 11.3% from ignition and 6.6% from decomposition (Table 3.5).

Combined. Total carbon emissions are found to be insignificantly decreasing at the rate of 19,838 Mg C, with a standard deviation of $\pm 38,553$ Mg C per annum (Table 3.4). Carbon emissions from mature forests accounted for 13,342,762 Mg biomass and represented a total of 82.1% of the total carbon emissions, with 25.7% from ignition and 56.4% (Table 3.5). Carbon emissions from regrowth forests accounted for 2,906,128 Mg biomass and represented 17.9% of the total carbon emissions, with 11.3% from ignition and 6.6% from decomposition.

DOES REGROWTH FOREST SEQUESTRATION OFFSET CARBON EMISSIONS ?

Transect 1. Regrowth forest biomass sequestration was larger than biomass emissions for nine of ten years, a net gain of 511,749 Mg biomass (Table 3.6).

Transect 2. Regrowth forest biomass sequestration did not surpass biomass emissions for every year, a net loss of 2,824,463 Mg biomass.

Transect 3. Regrowth forest biomass sequestration did not surpass biomass emissions for every year, a net loss of 7,228,041 Mg biomass.

Combined. Regrowth forest biomass sequestration did not surpass biomass emissions for every year, a net loss of 9,540,755 Mg biomass.

Table 3.6 Sequestration versus Emissions (Mg biomass) for 1989 to 1998 for Transect 1, Transect 2, Transect 3 and Combined transects.

Transect 1											
	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	Total
Regrowth Biomass Sequestered	146,635	163,216	126,113	157,854	72,340	97,605	134,112	92,527	109,137	99,231	1,198,770
Carbon Emitted	94,107	63,873	67,237	45,629	80,754	81,288	57,368	75,067	65,839	55,859	687,021
Difference	52,528	99,343	58,876	112,225	-8,413	16,316	76,743	17,460	43,298	43,372	511,749

Transect 2											
	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	Total
Regrowth Biomass Sequestered	58,446	76,604	-3,488	90,467	45,065	87,615	100,806	40,075	102,049	96,507	694,146
Carbon Emitted	364,617	297,876	378,515	350,462	359,347	234,055	242,605	342,886	434,711	513,534	3,518,608
Difference	-306,172	-221,272	-382,003	-259,996	-314,283	-146,439	-141,799	-302,811	-332,663	-417,026	-2,824,463

Transect 3											
	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	Total
Regrowth Biomass Sequestered	160,641	142,310	-117,117	130,773	-101,301	114,123	225,571	166,855	283,094	231,117	1,236,066
Carbon Emitted	834,598	774,255	1,151,270	1,136,120	1,193,417	1,060,181	541,053	612,764	603,761	556,688	8,464,108
Difference	-673,957	-631,946	-1,268,386	-1,005,347	-1,294,718	-946,059	-315,482	-445,908	-320,666	-325,571	-7,228,041

Combined transects											
	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	Total
Regrowth Biomass Sequestered	365,722	382,130	5,509	379,094	16,104	299,343	460,488	299,458	494,280	426,855	3,128,982
Carbon Emitted	1,293,322	1,136,004	1,597,022	1,532,211	1,633,518	1,375,524	841,026	1,030,717	1,104,311	1,126,080	12,669,737
Difference	-927,600	-753,875	-1,591,513	-1,153,117	-1,617,414	-1,076,181	-380,538	-731,259	-610,031	-699,225	-9,540,754

DISCUSSION

The three transects and their sum exhibit different behaviors dramatically over the course of the study. The model methodology has the capacity to differentiate dynamic landscapes that store (sink) and deplete total above-ground living biomass (source). Regrowth forest biomass significantly affected the measurements of total above-ground biomass, increasing carbon emissions overall by 21.7% (Table 3.5). Taking these results, models which extrapolate carbon budgets without accounting for land tenure are likely to underestimate carbon emissions and overestimate carbon stored in biological systems. Considering that the Amazon Basin contains 65% of the earth's tropical rain forests, with global deforestation being responsible for 15% of atmospheric carbon (WRI 1991), the underestimated effects of carbon emissions from regrowth forests will influence estimates of global carbon cycles.

PATTERNS OF LAND USE DYNAMICS

Each of the three transects displayed a unique patterns of regrowth biomass sequestration, mature forest biomass loss, and total above-ground biomass throughout the study. Each pattern is marked by different infrastructure development and human population densities. Human population pressures increase from the north to metropolitan Manaus and decrease south of the Amazon River.

Transect 3, Metropolitan Manaus and Vicinity

The development of human-dominated land has increased and can be characterized by the spatial patterns it exhibits. One predominant development pattern

occurs in lands between the Amazon River and Reservo Ducke where urban-suburban sprawl has consumed adjacent non-reserve forest lands. Reservo Ducke has resisted deforestation but border infringement is visually apparent from satellite imagery, presumably from direct harvest and escape fires (Gascon *et al.* 2001). Reservo Ducke has become surrounded by deforested lands making it the largest documented mature tropical rain forest fragment, 10,000 + ha.

South of the Amazon River large *faziendas* within a 'fishbone' road infrastructure exist with land turnover rates slightly longer than metropolitan Manaus. There is a riparian to road development pattern exhibited north and south of the Amazon; riparian settlements predate the roads that later extend to these previously remote settlements. Road development leads to subsequent deforestation of adjacent forests. The northwest corner of Transect 3 exhibits this land development pattern especially well. Riverside deforestation, a direct result of the cost-effective transportation of harvested timber *via* water, is noted elsewhere within the Brazilian Legal Amazon (Laurance *et al.* 2001, Verismo *et al.* 2002). While it is prevalent in the vicinity of Manaus, the patterns of riverside development predating road construction have received scant attention in the literature on deforestation.

Transect 3 has the highest deforestation rate averaging $2,218 \pm 93$ ha yr⁻¹ (mean, standard deviation). At the current rate of deforestation all mature forest stands will be cleared in 36.2 years from present or 31 years if Ducke Reserve remains protected. There is no doubt that this high rate of deforestation, 2.6% per year or -789,704 Mg biomass

yr^{-1} , is the primary reason for the loss of total above-ground biomass of 690,971 Mg biomass yr^{-1} . The effects of regrowth forest biomass sequestration rate, 97,001 Mg biomass per year, accounts for the differences in the rates.

Anthropogenically influenced lands have an average age of 4.36 in 1999, meaning that a typical 1 ha plot will sequester 21.03 Mg biomass before it is recommitted to slash and burn agriculture. Because of this quick turnover rate, regrowth forest biomass does not significantly influence the rate of decrease of total above-ground biomass (comparison of slopes test, $p = 0.07793$). Regrowth forests are still shown to significantly influence the quantity of biomass between 2.5 to 8.3%, (comparison of elevations test, $p < 0.0001$). Combustion and subsequent decay of regrowth forest biomass' represents 17.9% of the carbon emissions (Table 3.5). The contribution of regrowth forest biomass to the total carbon emissions has not previously been quantified, nor have these land use emissions been quantified relative to those of mature forest biomass combustion and decay.

Transect 2. Moderate Population Density With Increased Accessibility Due to Transportation Improvements

Directly north of Manaus, Transect 2 is characterized by the expansion of deforestation between the newly improved roads of AM 174 and AM 10. AM 174 extends to Caracas, Venezuela and is a significant trade corridor. AM 10 goes east to Itacoatiara and also handles considerable traffic. Recent road improvements in the western part of the transect are very likely to encourage new inhabitants and lead to a more rapid rate of deforestation. Once the available land is occupied, land values will

increase, especially for deforested lands. As deforested land value increases in response to population pressures, land tenure systems will decrease the fallow period. With cleared land being limited and land values increasing, economic incentives to further deforest adjacent lands will increase in the near future.

Transect 2 lost total above-ground biomass at the rate of 279,399 Mg biomass yr⁻¹ but this rate of decrease was significantly different from that of mature forest biomass which has decreased at the rate of 339,794 Mg yr⁻¹ (comparison of slopes test, $p < 0.0002$). Throughout the length of this study, regrowth forest biomass sequestration significantly increases the slope of total above-ground biomass. The ramifications of this finding are important in light of carbon models that have previously underestimated the influence of regrowth forest dynamics (Houghton *et al.* 1999). The average age of regrowth forests in Transect 2 is slightly higher than that of Transect 3, but is more variable fluctuating between 3.6 and 5.5 years. Because regrowth forest is limited in spatial extent and of a relatively young age, it only contributed to 14.3 % of the total carbon emissions. Transect 2 lost mature forest at the rate of 954 ± 102 ha yr⁻¹ (mean, standard deviation). At the present rate of deforestation, all mature forests will be compromised within 197 years.

Transect 1, Low Population Pressure with Relaxation of Development Pressures?

Further north, human population and development pressures subsided during the late eighties and mid-nineties. Large cattle *faziendas* were abandoned as transportation costs exceeded market value at a time when AM 174 was largely unimproved (*e.g.* dirt road). Fallow pastures succeeded into regrowth forest systems. The *Favienda Porto*

Alegro of BDFFP has several large holdings of regrowth forests exceeding 20 years of age (Figure 3.10). In 1995 AM 174 was improved to an asphalt surface, after which traffic between Manaus, Boa Vista and Venezeula increased considerably. Despite its close proximity to an improved road, however, the area has experienced little deforestation. Several large regrowth forest areas adjacent to AM 174 have had several rotations of agricultural use.

Transect 1 produced results that differed in direction and quantity relative to the other transects. Total above-ground biomass was shown to be increasing at a nominal rate of 80,946 Mg biomass yr⁻¹, while mature forest biomass was decreasing at the rate of 44,262 Mg yr⁻¹. The biomass sequestered by regrowth forests was on average 2.8 times greater than that lost by mature forest biomass. Accounting for regrowth forest biomass reversed the negative trajectory based upon mature forest biomass alone. The average age of a regrowth forest plot within transect 1 was 12.9 in 1999 and is increasing at the rate of 8.5 months per annum. It is worth noting that at this age, the majority of regrowth forest are sequestering carbon at a rate that is 35% less than younger regrowth stands exhibiting more vigorous growth (Figure 3.2).

Transect 1 typifies tenure systems that have an abundance of anthropogenically altered land relative to the development pressure exerted on these systems in areas more densely populated. It is a logical consequence of the extensive labor required to clear a mature forest; where an individual would much rather clear a regrowth forest with a machete in comparison to having to mechanically harvest a mature forest with chain saws, let the harvest dry over a period of several months, ignite the downed forest and then prepare and clear for the pasture or agro-system. Conversion of mature forest to

agriculture is prohibitive in terms of labor and capital investment. Typically, capital investment is not associated with landless people squatting on unoccupied land. Transect 1's relatively low deforestation rates ($124 \pm 34 \text{ ha yr}^{-1}$) indicates that tenants prefer to clear fallow regrowth stands. Regrowth forest carbon emissions represented over a third of the total produced, fully 53% higher than would have been predicted by models that only use mature forest deforestation as an estimate of carbon emissions (Houghton *et al.* 1999). Transect 1 is likely to typify the behavior of past trans-Amazon development projects that have had high rates of abandonment because of their inaccessibility to economic markets.

All Transects Combined

The results of the three transects were combined so that divergent behaviors could be evaluated quantitatively. Transects 3 and 2 lost total above-ground biomass while Transect 1 sequestered it. Total above-ground biomass was heavily weighted by the more populous southern transects, which showed a strong and steady loss. Since the rates of total above-ground biomass and mature forest biomass are significantly different, the contribution of regrowth forest biomass is important. This finding reinforces the importance of regrowth forest biomass and land tenures' influence on total above-ground biomass. Average regrowth forest age fluctuated between 3.5 and 6 years, demonstrating that the majority of deforested areas are short rotation slash and burn agriculture. Total carbon emissions were insignificantly decreasing, with regrowth forest emissions representing 18% of the total carbon emitted, meaning that the inclusion of regrowth forest emissions elevated total carbon emission estimates by 39% (Table 3.5).

THE AGE MAP MODEL'S BEHAVIOR IN REGROWTH FOREST BIOMASS

In the simplest scenario, a system will increase total above-ground biomass if deforestation is held to zero for one year. Total above-ground biomass would then increase at the same rate as regrowth forest biomass sequestration. Considering that each of the transects is made up of multiple and different land uses and land tenure systems, it is highly unlikely that deforestation would ever equal zero. A more realistic model would assess whether or not total above-ground biomass is increasing or decreasing by the ratio of mature forest biomass lost to the amount of biomass gained via regrowth forest sequestration. In the vicinity of Manaus, 39.1 ha of young sequestering regrowth forest ($9.1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) is needed to compensate for the biomass lost from each hectare of mature forest that is deforested (346 Mg ha^{-1}). For older regrowth forests, this ratio increases to 60.3: 1 since older regrowth forests sequester biomass at a slower rate ($5.9 \text{ Mg ha}^{-1} \text{ yr}^{-1}$).

At the other end of the regrowth forest age spectrum, we have the behavior of very young regrowth systems. These systems are highly variable in biomass composition relative to older regrowth systems (Steininger 2001). The high variability is due to the previous land tenure and the amplitude of this tenure (Uhl *et al.* 1988). Regrowth forest stands 0 to 2 years of age produced a negligible amount of sequestered carbon ($1.08 \text{ Mg biomass by year 2}$). In comparison to the ratios presented above, 330 ha of young regrowth forest sequestration is needed to compensate for the biomass contained within 1 ha of mature forest. If an area is comprised of predominantly young stands which were continuously cleared, the transect could theoretically have abundant regrowth areas with

nominal biomass sequestration. To quantify how prevalent very young regrowth stands are at the end of the study, the 1999 age map was queried to generate descriptive statistics of this less contributive group. The majority of Transect 3's regrowth forests are less than 2 years of age (52.7%). Transect 2 has 42.1% of its regrowth being 2-years-old or younger. Conversely, Transect 1 has only 11.6% being 2 years of age or less. With all transects combined, the 2-year-old or less age class represented 43.4% of all regrowth forest systems. It should be acknowledged that each year will produce different age class distributions based upon that year's tenure systems and aberrant natural events (in 1997-98 for example, El Nino produced dramatic fires throughout the Amazon Basin). The predominance of these young regrowth systems is also represented by the relatively younger average ages contained in Transects 3 and 2.

REGROWTH FOREST BIOMASS SEQUESTRATION *VERSUS* CARBON EMISSIONS

The controversy over whether or not the biomass of regrowth forest sequestration compensates for mature forest biomass lost ties in with the above analysis of total above-ground biomass, with the exception that regrowth and mature forest emissions have an initial pulse of carbon release followed by a negative exponential lagging tail of decay. Typically, studies recognize that this tail extends for ten years when most of the carbon has been volatilized (Fernside 1992) but still accounts for the decay within a single year's record. Our study is unique in that it tracks regrowth forest combustion and decay as well as that of mature forest combustion and decay. It also uses different equations for regrowth and mature forest based upon the persistence of biomass in anthropogenically

altered landscape within the Amazon Basin through time (Fernside 1992, Fernside *et al.* 1993, Steininger in review).

Transect 1's regrowth forests did sequester more carbon than the total carbon emitted for nine of the ten years in this study. Typically regrowth forest sequestration exceeded carbon emissions by 74% in this sink habitat. Sink habitats within the Amazon Basin are not well documented (but Phillips *et al.* 1998 have proposed that mature forests are indeed sink habitats, and others have found that liana biomass is significantly increasing in the Pan-Amazon periphery, Phillips *et al.* 2002), especially within anthropogenically modified habitats. Within this transect, most of the large-scale cattle *faviendas* were abandoned in the late 1980's and subsequently never redeveloped. During the 1980's the main thoroughfare, AM 174, was comprised of a dirt road that sometime restricted vehicular speed to 15 km hr⁻¹. Without federal and state government incentives, the transportation costs of getting cattle to market were prohibitive. As previously noted, a recent increase in land reoccupation has occurred in the late 1990's, presumably associated with road improvement of AM 174. Such improvements greatly decrease the cost and time needed to get produce to markets and are expected to dramatically increase the scale of deforestation throughout the Brazilian Amazon (Laurance *et al.* 2001, Nepstad *et al.* 2002, Bergen *et al.* in Chapter Four).

The age model is robust and identifies the differences between the three transects. These divergent trajectories occurring at smaller scale can be quantified and qualified by summing all transects. The southern transects are considered consistent sources of atmospheric carbon because total carbon emissions always exceed carbon sequestered. Within Transect 2 annual carbon emissions exceeded annual regrowth sequestration by a

ratio of five to one. Within Transect 3 annual carbon emission exceeded annual regrowth forest sequestration by nearly seven to one (6.8: 1). With the three transects combined carbon emissions exceeded sequestration by a factor of four to one. One explanation for the dramatic increase in this ratio comes from the age model itself. Previous attempts at quantifying carbon emissions subscribed to a less complex form of carbon bookkeeping, which assumes regrowth forests are collectively static and contribute the same amount on an annual basis, regardless of the tenure system and its amplitude. Contributing further to this inaccuracy, studies typically only use analyses that occur between two years, and then extrapolate these findings into the future. With two dates of satellite imagery separated by 10 years, lands that were initially presented as regrowth forest and terminated in regrowth forest are assumed to have sequestered carbon at a constant, steady annual increment. In reality these lands might well have been cleared subsequent to the initial imagery, maintained as open pasture and then allowed to go fallow (physically possible and visually apparent in all three transects conducted in this study). The difference between the amount of carbon sequestered would be overestimated (54.6 *versus* 9.2 ha⁻¹) and the failure to account for regrowth forest emissions would result in a miscalculation by at least 45.7 Mg biomass ha⁻¹. Thus land tenure dynamics must be taken into account to obtain accurate and precise measurements of total above-ground biomass. By contrast, the carbon accounting models ignore the economic efficiency of swidden agriculture systems. This is not meant to infer that people do not deforest areas in order to establish agriculture. In fact, INPE's 2000 report quantifying deforestation within the Brazilian Legal Amazon assigns the majority of deforestation responsibility to swidden agricultural systems. The greater value of deforested land places regrowth

forests in higher demand, leading to shorter rotation periods than previous carbon balance studies have been methodologically capable of recognizing.

PRECISION, ACCURACY AND PAN-AMAZON EXTRAPOLATIONS

Accounting for land tenure systems in regrowth forests almost doubles the carbon emission estimates of the group and decreases the amount of regrowth forest biomass sequestration and total above-ground biomass. Any extrapolation across the Amazon Basin that does not account for land tenure will introduce inaccuracy and imprecision. If pilot studies are conducted within each of the state political regions, using the methodology in this study the accuracy and precision of these carbon budget models will be increased. Pilot studies should take full advantages of previous pan-Amazon surveys such as RADAM (1976), the current international funded Brazilian leading Large-Scale Biosphere-Atmosphere Experiment (LBA) and any areas that have complementary ground-based biomass estimations. MSS, TM, ETM+ satellite imagery have surveyed all areas within the Amazon on a biweekly basis since 1972. Although the majority of this imagery is unusable due to cloud cover, cloud free data are available for temporal studies similar to ours in Manaus. After pilot areas within individual states have been quantified, it would be more appropriate to extend these findings across the basin within a spatially explicit model format that is capable of distinguishing human land tenure systems amplitude, frequency, and effect.

Undoubtedly, estimates of total carbon emissions will increase and regrowth forest sequestration and total above-ground biomass estimates will decrease once their dynamics are included in a large-scale model. There is great potential for such large-

scale models. There are probably substantial areas within the Amazon that are sequestering biomass at significant rates (sinks). If these sink habitats were created previous to 1990 they qualify to be traded as carbon credits within domestic and international carbon markets. This should be seen as an economic incentive since most of the sinks would be abandoned agricultural land that is insolvent. Increasing this incentive is the fact that abandoned lands are usually quite cheap to rent and the regrowth forest that sequester upon them have moderately high sequestration rates making them among the most profitable regions for biomass sequestration in emerging carbon markets.

A number of items prohibit this study from being extrapolated across the Amazon Basin. First, the area studied is quite small relative to the rest of the forested Brazilian Legal Amazon (6,800 versus 480,000,000 ha, roughly 0.0014%). Another confounding factor is that the Amazon basin contains a diversity of distinct land use tenure systems. The large cattle *fazendas* and peasant agriculture systems typifying the eastern 'arc of destruction' are very different from the impact of *cobloco* rubber tapping and agro-forestry practices being implemented in the southwestern state of Acre. Corporate and wildcat mining operations in the states of Para and Roraima differ significantly in comparison to agriculture and livestock based tenure systems. Santarem and Belem are river towns associated with river commerce and natural resource extraction but differ by order of magnitude in terms of exportation from the international free trade zone that is Manaus. Concurrently, the rapid dynamics of Manaus are substantially different from parts of Amazonas that are more remote and accessible only by river or air transportation. Furthermore, as noted previously by Browder and Godfrey (1997), the individual states

that comprise the Brazilian Legal Amazon differ substantially in history, land use, land tenure as well as how federal government subsidies are spent.

CONCLUSION

This study documents the land tenure, fire, and deforestation occurring in the Manaus region from multi-temporal Landsat satellite imagery. The differences observed with different transects highlight the ability of this methodology to distinguish differences caused by land tenure, fire, and deforestation. This model is unique in that it specifically tracks each pixel's history, biomass accumulation, and ignition through time. By tracking the spatially explicit behaviors, we are able to more accurately estimate the contribution of secondary forests to total above-ground biomass. This finding has dramatic effects upon the measure of total above-ground biomass. In one transect studied with long turnover periods, carbon sequestration exceeded carbon emissions from mature and regrowth forest ignition and decomposition. Where traditional models considering just forest/non-forest classes would have estimated TAGB to be decreasing over time, this model demonstrates that TAGB is increasing. The ability to detect and identify carbon sinks, although technologically exciting, should not ease concerns over the Amazon's tropical deforestation contribution to global atmospheric carbon. In the majority of transects, as well as when all transects are combined, carbon emission exceeded sequestration. When the entire study area is accounted for emissions exceeded sequestration by over 9×10^6 Mg during 1999 alone. Although it might be tempting to extrapolate the results from this model across a pan-Amazon region, the spatial extent of the study limits its ability to make such projections. Within just the Manaus region the total above-ground biomass sequestered in remote areas while areas of higher human population pressure continued to deplete total above-ground biomass. To try to estimate how long stretches of the trans-Amazon highway that have been abandoned will

encounter rapid deforestation and escape fires more prevalent in east and southeastern portions of the Brazilian Amazon is both a spatial and temporal problem. A larger scale study based on more intensely documented areas will produce models of total above-ground biomass with an even higher accuracy and precision. A successful study of this scale would answer many questions concerning the Amazon basin's contribution to global atmospheric processes, as well as describe land tenure phenomena that are not prevalent in the vicinity of Manaus.

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Causes and Prevention of Habitat Degradation, Forest Fragmentation and Deforestation in the Brazilian Legal Amazon

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ABSTRACT

Laurence *et al.* (2001a) predicted that by the year 2020 only 45 to 5% of the Brazilian Legal Amazon will remain in a pristine forest condition as a result of the combined effects of large-scale development projects within the Brazilian Legal Amazon under a federal program called *Avança Brasil*. Many of the projects will dissect eastern Amazonia into smaller forests that will be greatly penetrated by western economic, non-endemic influences. Laurence *et al.* (2001a) included multiple sources of forest degradation but did not evaluate the individual contributions of infrastructural project types to forest habitat degradation. Since many infrastructure projects spatially overlap in close proximity to one another and are synergistic in their behaviors, parsing accurate estimates of each project types' contribution to forest degradation in the Brazilian Amazon is more complex and requires a rigorous methodology. To meet these needs, we developed Element Added and Element Deleted models that can parameterize the extent of a degradation that is associated with each type of development, *e.g.* habitat degradation associated with road upgrades *versus* that occurring from hydroelectric projects. Using a previously published Optimistic model (Laurence *et al.* 2001a), we find that within *Avança Brasil* (which has been recently renamed *Pluri-annual 2004 – 2007*), road upgrades followed by utility corridors, hydroelectric, railroads and river channelization are the most destructive infrastructure types within *Avança Brasil*. Road upgrades are also responsible for the majority of pristine forest degradation followed by utility corridor construction, new railroads, hydroelectric installations and river channelization. Beyond *Avança Brasil* projects, riverside deforestation is posed to have the greatest potential threat if it is fully capitalized. Present reserve systems prevent 151,777 km² of heavy

degradation and 277,269 km² of moderate habitat degradation from occurring. The deleterious effects of *Avança Brasil* projects can be decreased substantially if separate projects are engineered to place projects that occur in a region into narrow development corridors, locate utility corridors along present road infrastructure and design 'insulation' reserves in areas that require cross-corridor transit. Insulation reserves are more economically efficient, require less maintenance, and will prevent the penetration of previously undisturbed large forest tracts relative to development projects as they currently stand.

INTRODUCTION

Scientists are interested in tropical rain forest systems because these forests contain an inordinate diversity of life forms, complex food chains, elaborately intertwined multiple species life history strategies (bio-complexity), and they have a disproportionate influence on atmospheric greenhouse gas recycling and emissions. The Amazon tropical rain forest is the largest continuous rain forest in the world and have been estimated to house between 25 – 30% of Earth's total biological species diversity (Erwin 1983, Wilson 2000, The Nature Conservancy *pers comm.*). At present the *Brazilian Legal Amazon contains 40% of the world's remaining rain forest (Laurance et al. 2001b)* and has come into increased examination since its high potential for global greenhouse gas emissions from the conversion of dense high carbon forest to low carbon swidden agriculture and pasture systems. These forests sequester carbon (CO₂) from the atmosphere at faster rates than temperate and boreal forest systems. The balance between carbon emissions and sequestration in Amazon forests has a dramatic influence on global bio-geochemical systems and the rate of global warming (World Resources Inst. 1990, Myers 1991, Houghton *et al.* 1999, Defries *et al.* 2002, Bergen and Steininger in Chapter Three). Thus, increasing our understanding about the rates of deforestation in the Amazon will help advance our knowledge of the global geochemical cycles. However, the effects of deforestation in the Brazilian Amazon go well beyond the conversion of forest to pasture and must include the effects of infrastructure development that indirectly accelerate forest conversion. This issue is of current interest as the Brazilian government *plans to expand infrastructure in the Amazon that will increase deforestation over the next 20 years through government sponsored projects via international lending programs.*

Under proposed programs, the expansion of road building and improvement, utility corridors, hydroelectric projects, railroads and river channelization will penetrate previous undisturbed large forest tracts in southern and southwestern Amazonia (Carvalho *et al.* 2001, Laurance *et al.* 2001a, Nepstad *et al.* 2001a).

Skole and Tucker (1993) conducted one of the first analyses quantifying Brazilian deforestation rates. This study examined remotely sensed 1978 Multi-Spectral Scanner (MSS) and 1988 Landsat 5 Thematic Mapper (TM) satellite imagery for 85% of the Brazilian Legal Amazon. They reported the Amazon is being deforested at a rate of 15,000 km² per year. This was much less than previous estimates of 50,000 to 80,000 km² (Myers 1991, World Resources Inst. 1990). Of particular importance were the total areas affected by habitat fragmentation and forest degradation, which were estimated to be an additional 38,000 km² per year as quantified by a geographic information system (GIS) buffer analysis.

An important contributor to Amazon deforestation is fire, both accidental and intentional. Nepstad *et al.* (1998) delineated areas that are susceptible to fire in forested and non-forested regions of Brazilian Amazonia. They quantified an additional 257,238 km² of tropical forest that had a high fire potential resulting from increased fragmentation, edge effect, direct deforestation, and dramatic seasonal variations in rainfall. An additional 161,719 km² of forest were at a moderate risk of fire. The 1997 - 1998 El Nino dramatically decreased the total rainfall across the Amazon Basin. Associated forest fires were so prevalent that airport traffic was halted due to poor visibility from smoke on several occasions during the El Nino.

Fire processes also affect the intensity of subsequent accidental fires. Cochrane *et al.* (1999) studied ground measurements of fire processes in the eastern Brazilian Amazon. They found that initial ground fires with small flame lengths kill 95% of the trees with a diameter breast height (DBH) of less than 20 cm. Larger trees survived until larger secondary ground fires erupted killing 98% of all trees regardless of DBH. In these secondary fires, greater flame length incidence time occurred because of increased fuel load from dead trees of the primary ground fire, which resulted in the death of larger trees. Within the 'arc of deforestation', forest fires affected 50% of forests. The primary source of ignition was escape fires from adjacent slash and burn agriculture. Accidental fire increased the deforestation estimates by 129% for the two years of study. This study further quantified that 90% of accidental fire deforestation occurred during *El Nino* years. Thus, the deforestation effects of slash and burn fires are much larger than areas targeted for conversion to agriculture.

Nepstad *et al.* (1999) studied a larger area within the arc of deforestation and had similar conclusions about fire and logging in Brazilian Amazon, mainly that the effects are under-reported. They found that 10,000 to 15,000 km² yr⁻¹ of forest is logged and not accounted for in traditional forest/ non-forest estimates produced by MSS and TM satellite data. The study supports the finding that traditional satellite classification misses 50% of the area that is logged each year. In the region of Paragominas, Pará, in an area which TM data estimates as 62% forested, had only 10% of its forest unadulterated by logging or fire. Further, aerial transects quantified that 9% of forested areas had sustained recent fire activity.

Nepstad *et al.* (2001a) expanded their findings on logging and fire to the entire Legal Brazilian Amazon. They found that if 6,245 km in road improvements were to take place within the Brazilian Amazon. This will drastically increase the likelihood that forests 'very prone' to forest fires will have a greater potential to ignite from road sources. It is worth acknowledging that of the 6,245 km of roads to be improved, no less than 4,329 km already exist in an improved paved state (Appendix C, Bergen unpublished data). They further argue that one third of Brazilian Amazon forest is susceptible to fire during severe *El Nino* episodes. By their estimates, 120,000 - 270,000 km² of additional deforestation and degradation via fire and logging would occur. Fires are thought to perpetuate other fires through three positive feedback loops: (1) fires promote drought, (2) fires increase the susceptibility of forests to recurrent burning, and (3) fires self perpetuate by burning agriculture and forest systems. Individual and community based fire management would result in less forest and agriculture being destroyed by escape fires. If fire were better managed, it would lead to a higher economic efficiency for agricultural systems.

In 1999, the Brazilian Government sought \$ 40 billion to fund infrastructural development in a program called *Avana Brasil* (renamed *Pluri-annual 2004 - 2007*). To examine the effects of this project on deforestation rates in the Brazilian Amazon, Laurance *et al.* (2001a) produced two spatially explicit models that took into account all of the *Avana Brasil* projects plus fire, new road building, natural resource extraction and present reserve configurations. These models, named "Non-optimistic" and "Optimistic", found that 27 to 40% of the Brazilian Amazon will be deforested or heavily degraded by 2020. *Avana Brasil* was shown to increase forest fragmentation by 36%. Much of the

forests that exist in eastern and southern Amazonia will be hit the hardest and the remaining large contiguous forest blocks in central and northern regions are likely to be fragmented into much smaller sized blocks from infrastructural penetrations into formerly remote forest. Under the Non-optimistic scenario, the remaining pristine forest blocks will occur only in western Amazonia in the year 2020. Nepstad *et al.* (2001b) critically stated that this modeling approach included too many infrastructure projects to evaluate each infrastructure type's influence on the forested Brazilian Amazon.

The goal of this study is to elaborate how each of the elements that comprise the Laurence *et al.* (2001a) optimistic model affects deforestation rates, how the model is constructed from these diverse factors, and how this methodology is calibrated with remotely sensed data. Despite the models criticisms of interpretability (Nepstad *et al.* 2001b), a methodology is constructed that can estimate the effects of development types across the Brazilian Legal Amazon. It is hoped that these data will help focus current efforts to decrease the rate of deforestation and fragmentation within the Brazilian Legal Amazon. We further suggest management strategies to ameliorate these development's deleterious indirect effects upon this diverse and bio-complex region that influences global bio-geochemical cycles.

METHODOLOGY

The methodology used to construct the Full and Null models in this study is identical to that published previously for the optimistic model (Laurance *et al.* 2001a). Because of the conciseness of the initial study, the methodology will detail the models construction and show how the model quantifies deforestation and habitat degradation. In several instances the primary spatial datasets have been updated to respond to changes of *Avana Brasil*'s development projects and reserve systems. The 'optimistic' model of Laurance *et al.* (2001a) is chosen for further investigation because its results are more *conservative and consistent (within 5%) with other land use projections occurring within the Brazilian Legal Amazon* (Nepstad *et al.* 2001a, INPE 2002). This does not mean that development issues within the Brazilian Legal Amazon have abated or decreased. Recent reports quantify that the rate of deforestation is increasing (Laurance *et al.* 2002a). The optimistic model is used simply because its findings are only slightly less than previously published studies.

The description of the model construction is broken down into several key steps. The first step describes the model elements, why they were chosen and a brief description of their effects upon forested systems. Each of these model elements has a different spatially explicit behavior that is projected to the year 2020. These projections are taken from estimates of deforestation from previous projects or data gathered for this models construction. These model elements are then coded and combined across the Brazilian Legal Amazon *via* GIS. Since some of the model elements react synergistically with *other elements, a more comprehensive analysis is presented to quantify the element's* spatial influence throughout the study area.

MODEL ELEMENT'S SPATIAL BEHAVIOR

The model elements are used to predict the future forest conditions in the year 2020. To do this, we classified forest structure into four broad land-use categories. Each of the model elements has a unique behavior that is either investigated here, supported by previous scientific study, or given by expert's best approximation.

Our models predict the spatial distribution of four land-use categories: (1) heavy impact areas that have primary forest cover that is absent or markedly reduced, and heavily fragmented; such areas are highly vulnerable to edge effects, fires, logging, and over hunting, and are severely degraded ecologically, (2) moderate impact areas that have mostly intact primary forest cover ($> 85\%$) but contain localized forest clearings and some roads, and may be affected by logging, mining, hunting, and oil and gas exploration, (3) light impact areas that have nearly intact primary forest cover ($> 95\%$) but can experience illegal gold-mining, small-scale farming, hunting, hand logging, and non-timber resource extraction (*e.g.*, rubber-tapping), and (4) pristine areas which have fully intact primary forest cover and are free from anthropogenic impacts aside from non-westernized forms of hunting, fishing, and swidden farming by traditional indigenous communities.

MODEL ELEMENTS

The model is constructed from the overlay of 17 spatially explicit modeling elements chosen for their known effects on deforestation or inclusion as an infrastructure development project within the Brazilian Legal Amazon. Some elements are development projects under *Avança Brasil*, others are existing infrastructure that have

influence on deforestation rates within the Amazon, such as reserves and/ or delineated indigenous lands. The elements within *Avanca Brasil* include road improvements, hydroelectric projects (completed before 2011), new railroads, utility corridors, and river channelization. Elements that are important for predicting deforestation but not included in *Avanca Brasil* include timber concessions, mining, late schedule dams (completed after 2011), new road construction, fire potential, and riverside deforestation. If an element is to be expanded in the future, it is subdivided into its present and future locations. Several model elements could not be explicitly projected into the future. Their analysis is confined to their current spatial distributions.

Road Development and Expansion

Roads are divided into two subcategories, improved and unimproved roads. Improved roads are paved and capable of commercial traffic. Unimproved roads vary in their individual capacity for vehicular traffic. Many unimproved roads are seasonal and have conditions that change rapidly. Both road categories are to be expanded either by federal, state, or private financiers (Appendix C).

Currently there are 4,329 km of improved roads existing in the Amazon. Under proposed projects, there are 2,955 km of new road improvements that are to occur on previously unimproved roads. The total improved road network will be 7,284 km. The current distribution of unimproved roads covers at least 46,375 km with plans for 1,275 km of new roads bringing this section of the road network to 47,650 km.

To predict the impacts of highways and roads planned for the next 20 years, we assessed the effects of existing highways and roads on primary-forest cover during a

recent 15 to 25 year period. We overlayed the 1995 road network on the TM based Pathfinder map of the Brazilian Amazon for 1992 (<http://www.bsrsi.msu.edu>). Many of the region's major highways (*e.g.*, Belém-Brasília, Transamazon Highway, BR-364) were constructed in the 1960s and 1970s, and thus had been in existence for 15 to 25 years by 1992, comparable to the 20-year time frame used in our predictions. Initially, five 'investigation zones' were created around all paved highways (0 to 10, 11 to 25, 26 to 50, 51 to 75, and 76 to 100 km on each side of the highway), and the percentage loss of primary forest cover within each zone was quantified. This analysis was then repeated *using the entire network of highways and unpaved roads*. Clouds, cloud shadows, savanna and rivers were excluded from the analysis. Areas beyond the Brazilian Legal Amazon were excluded from the calculations. Deforestation was registered only for closed canopy forests; losses of other habitats were not included.

The analysis (Figure 4.1) revealed that deforestation is strongly increased near highways and roads. Both roads and highways averaged about 30% forest loss within the 0 to 10 km zone, but highways had more far-reaching effects than roads, averaging about 20% and 15% forest loss in the 11 to 25 and 26 to 50 km zones, respectively. Roads tended to cause more localized deforestation, with average forest loss declining below 15% for areas further than 25 km from the road. The most far reaching effects we observed were the construction of 200 to 300 km long state and local roads projecting out laterally from several major highways. Since this calibration methodology also measures deforestation that could occur from other development sources adjacent to roads, it has *the potential to overestimate deforestation caused by roads*.

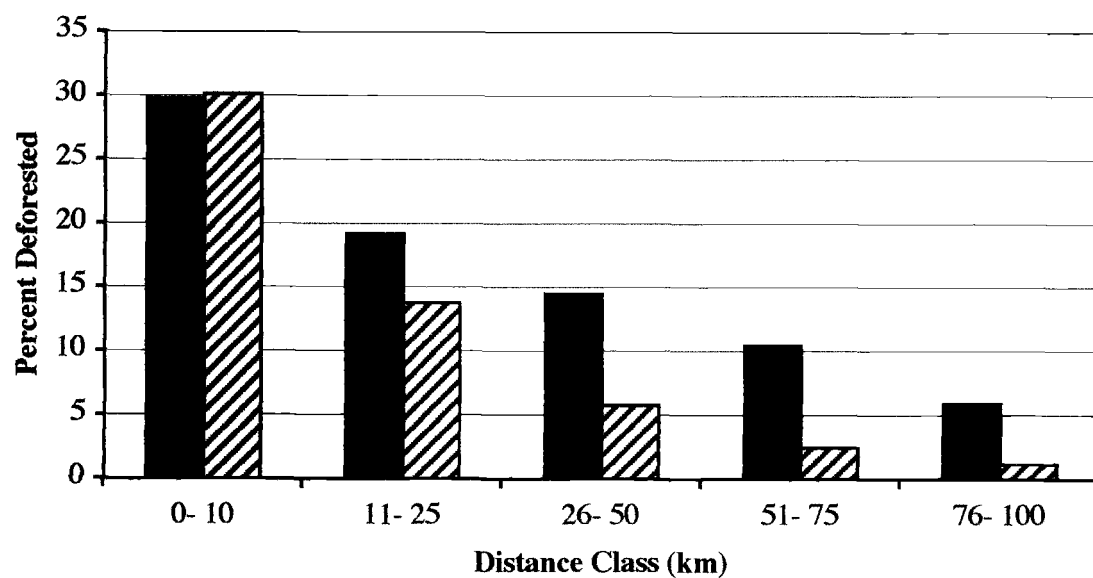


Figure 4.1 The percentage of land deforested adjacent to improved (solid) and unimproved roads (diagonals) for the entire Brazilian Legal Amazon in 1992.

Forests adjacent to improved paved highways are classified into the four landuse classes mentioned. Areas within 75 km of highways are classified as lightly degraded, areas within 50 km are classified as moderately degraded and areas within 25 km are classified as heavily degraded.

The impacts of unimproved roads are less destructive than those of improved paved highways. Forested areas adjacent to unimproved roadways are classified accordingly with forest areas within 50 km being classified as lightly degraded, forested areas within 25 km as moderately degraded, and forested areas within 10 km being heavily degraded.

Hydroelectric Projects

There are two large hydroelectric projects in operation within the Brazilian Legal Amazon, *Balbina* and *Tucuruí*. These projects have very different deforestation trajectories presumably due to the drastically different regional contexts and levels of protection from reserves (Figure 4.2).

Tucuruí exists in the heart of the 'arc of deforestation.' Most of its electricity production is consumed by an aluminum producing facility (Browder & Godfrey 1997). The plant provides moderate income to its workers as well as notable accessory economic incentives. The *Tocantins*'s inundation displaced several townships that were moved from the riverbanks to higher ground (further increasing deforestation, *Tucuruí* hydroelectric dam is on the *Tocantins* river). The *Tucuruí* hydroelectric project has been well researched and effects of deforestation have been quantified (Browder & Godfrey 1997).

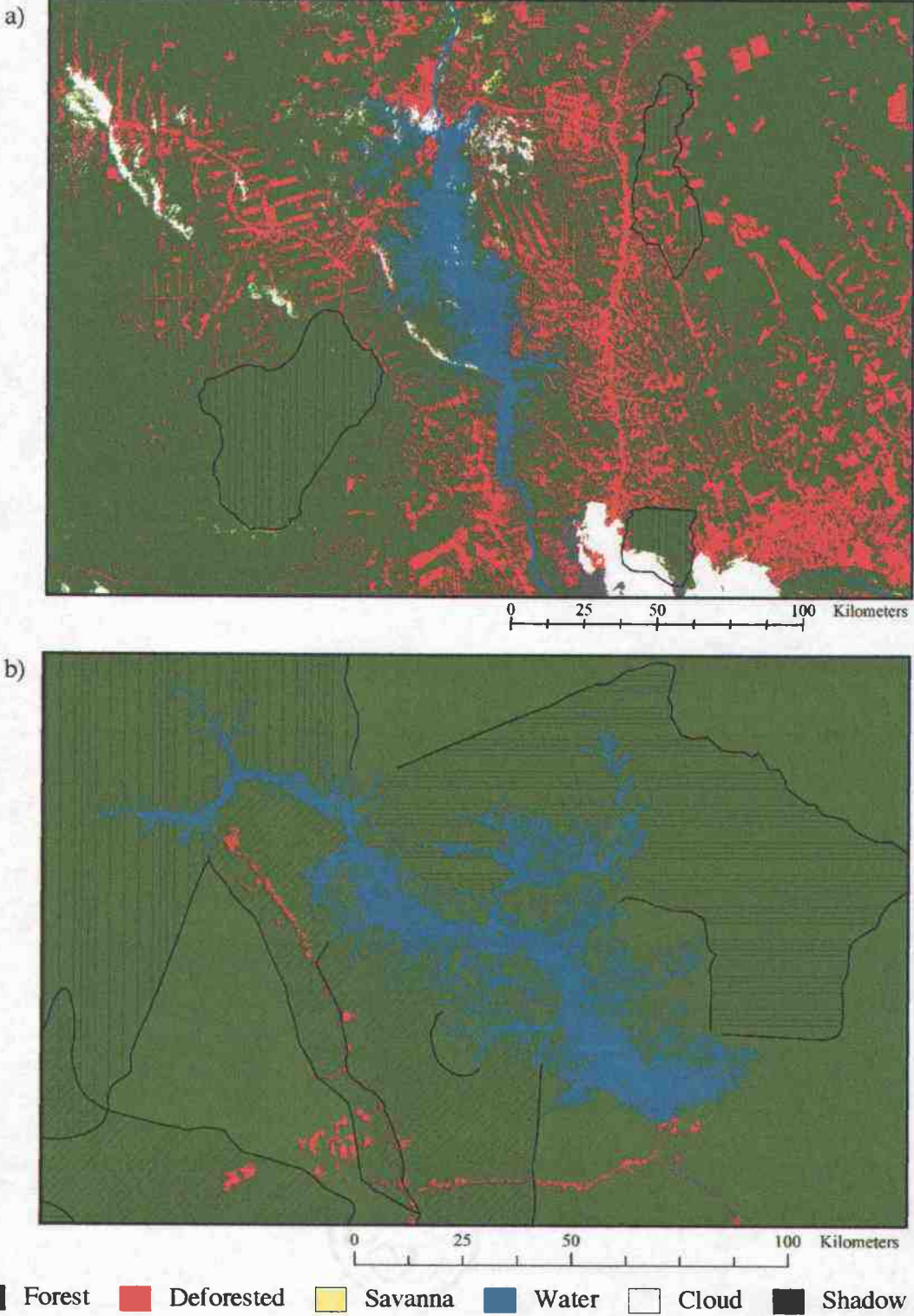


Figure 4.2 Comparison of the Tucuru (a) and Balbina Hydroelectric projects (b). Reserves are delineated with bars for high protection, diagonals for moderate protection and stripes for indigenous lands.

The *Balbina* hydroelectric project operates approximately 110 km north of the city of *Manaus*. The majority of electricity produced is consumed by the city of *Manaus* (an economic free trade zone with multi-national assembly plants). *Balbina* has deforestation to the southwest towards the town of *Presidente Figueiro*. The deforestation in this area is associated with access roads to the dam and the establishment of large-scale agricultural farms. The majority of the remaining shoreline is surrounded by a complex of indigenous lands (*Waimiri - Atroari*), moderate protection of sustainability reserves (*Apaes Caverna do Moroaga*), and high protection ecological reserves (*Reibo do Uatuma*). The deforestation occurring at *Balbina* is less severe relative to that incurred at *Tucuruí*, most likely due to the extensive reserve systems and distance from economic markets.

Avança Brasil proposes several additional hydroelectric projects centered on the upper *Tocantim* and *Xingu* watersheds (Appendix A). Hydroelectric projects slated for completion before 2011 are classified as near future dams, dams slated to be developed beyond 2011, not part of *Avança Brasil*, are designated as late development hydroelectric projects and are expected to be completed by the year 2021.

We decided to take a conservative approach to modeling the deforestation associated with these new hydroelectric projects. Forested areas within 10 km of the projected shoreline are placed in the lightly degraded class. Forested areas within 5 km classified as moderately degraded. Areas to be inundated are classified as heavily degraded. This is consistent with the more recent environmental regulations that the federal government has in place, i.e. *Balbina*.

Channelization

Major channelization projects are intended for the *Rio Madeira*, *Rio Tocantins* and *Rio Araguaia* that will open these areas to barge traffic (Appendix A). No channelization projects presently exist within the Amazon Basin. River channelization projects seek to alter the seasonality and depth profile of seasonally fluctuating rivers. By maintaining consistent water depths for barge and boat traffic, seasonal pulses in water flow will be disrupted. This has the potential to disrupt seed distribution of aquatically pulsed *várzea* forests as well as significantly alter fish species distribution patterns occurring within the rivers. Many frugivorous fish species are tuned to these seasonal pulses that occur when trees set fruit (Smith 1981, Goulding *et al.* 1996). Forest and fisheries have been significantly altered in both the Tucuruí and Balbina hydroelectric installations requiring the government to supply training, food, and health services for groups that have become alienated from previously reliable protein sources (for example, in Balbina the federal government has a 25 year center opened for the *Waimiri-Atroari*, *Posto FUNAI*).

Channelization projects will undoubtedly have numerous maintenance roads and infrastructure to service the new commerce corridors. Channelization projects have not penetrated into the Legal Amazon so their effects on forests are hard to substantiate. Since roads will be included during and after the creation of these projects, we have used similar deforestation characteristics. To model the impacts of these river channels forested areas within 50 km are classified as lightly degraded. Forested areas within 25 km are classified as moderately degraded and forested areas within 10 km are classified as heavily degraded.

Railroads

Railroads have existed since the late 19th century in Rondonia. There are other smaller railroads in Maranhão and Amapá. Proposed railroad projects seek to extend rail service from Porto Velho and Santarém to central economic markets in Brazil (Appendix C). These railroads will make it cheaper to transport timber (Nepstad *et al.* 2002a, Veríssimo *et al.* 2001) and agricultural products to global markets. Brazil is the second largest exporter of soybeans and is expected to surpass U.S. soybean production by the year 2010. Recently Brazil has lifted the ban on importing genetically modified crops. *Monsanto® has patented genetically modified soybean cultivars capable of yielding crop* in southern Amazonia. The threat of the spread of cash crops into the southern Amazon has been cited as a major new threat to forested systems within the Amazon (Nepstad *et al.* 2002a). To model the impacts of the new railroads forest areas within 50 km are classified as lightly degraded. Forested areas within 25 km are classified as moderately degraded. Forested areas within 10 km will be more impacted and are classified as heavily degraded.

Utility Corridors

Major utility corridors are proposed to export electricity and natural gas pipelines beyond the Brazilian Amazon (Urucu to Manaus *via* Coari and to Porto Valho, Appendix C). Electrical corridors are needed to transfer electricity generated from hydroelectric projects to other regions of Brazil, Bolivia, and Venezuela. Electricity is in high demand and Brazil had major power shortfalls in 2001. Natural gas is also in high demand since Brazil lacks large indigenous sources of fossil fuels. The connection of the Urucu natural

gas fields to Manaus and Port Velho will greatly expand infrastructural developments into the previously undeveloped western Amazon.

Utility corridors will have numerous maintenance roads. Since roads will be included during and after the creation of these projects, we have used similar deforestation characteristics for these corridors. To model the impacts of these infrastructure projects forested areas within 10 km of utility corridors are classified as heavily degraded. Forested areas 10 – 25 km from utility lines are classified as moderately degraded. Forested areas that are 25 – 50 km from the utility corridors are classified as *lightly degraded*.

Timber Concessions

Logging within the Brazilian Legal Amazon is a significant source of deforestation (Skole and Tucker 1993, Nepstad *et al.* 1999, Laurance *et al.* 2001b). Illegal and legal timber harvests are considered within the model. Brazil has made substantial efforts to curb the sale of illegal timber harvests within the Amazon. This study uses the maps created by IMBAMA studies (1998, Brazil's national environmental agency, Appendix C).

In addition to directly damaging forests and aquatic ecosystems, logging activities create road and skid-trail networks that increase access to forests for slash and burn farmers, ranchers, and hunters. The Amazon is becoming an increasingly important source of tropical timber. Brazil has nearly 400 domestic Amazonian timber companies, *and there has been an influx of multi-national timber corporations from Asia*. Asian corporations invested over \$ 500 million in the Brazilian timber industry in 1996 alone,

and currently control at least 13 million ha of Amazonian forest (Laurance *et al.* 1998, Laurance *et al.* 2001b).

Increased human intrusion into these forests *via* skid trails and logging roads will undoubtedly increase hunting activities that are commonly associated with timber operations within the Amazon and worldwide (Robinson *et al.* 1999, Laurance 2001, Fa *et al.* 2002, Peres and Lake 2003). Increased hunting pressures are very likely to cause trophic cascades that ultimately change forest composition and dynamic (Terborgh 1988, Terborgh *et al.* 2001). Despite present selective cut forestry practices, areas that are subject to logging are classified as moderately degraded.

Mining

Illegal mining for gold by *garimpeiros* is rampant in parts of the Brazilian Legal Amazon and has been studied in socio-economic terms (Browder & Godfrey 1997, *garimpeiros* is Portuguese for gold miner). A 1998 map of estimated legal and wildcat gold mining and industrial mining, produced by IBAMA, is used for this study (Appendix C). Legal mining is closely monitored by Brazilian federal agencies and has been benign in its habitat destruction when environmental concerns are addressed prior to mine construction.

Mineral resources (iron ore, bauxite, gold, copper) provide a rapidly growing economic impetus for road building in the Amazon (Browder & Godfrey 1997). Legal mining operations are highly restricted in their use of and access to adjacent forests. Wildcat mining, typically associated with gold mining, occurs on a relatively small spatial scale. Although these operations can be locally devastating to forests, they are

usually confined to small areas and near riparian strips (water is needed to uncover soil to expose mineral material). Forested areas exposed to mining are determined to be lightly degraded.

High Protection Reserves

Ecological high protection reserves are the combination of national and state parks, ecological reserves, biological reserves and ecological stations within the Brazilian Legal Amazon (Appendix C, Table C.1). The northwestern border of the Brazilian Legal Amazon is a network of high protection federal reserves, moderate protection sustainability reserves and indigenous lands that are spatially intermixed (Appendix C). Some locations within this area are within multiple reserve systems and delineated indigenous lands. Considering that these intermixed reserve systems exist along the 1,000 km+ border with Venezuela and Columbia and that reserve parcels exist in numerous reserve holdings, the total amount of reserves is not a simple additive function.

The reserves also act to negate deforestation and degradation associated with other factors that might be occurring in proximity to the reserve. High protection reserves are assumed to have active management to prevent natural resource exploitation within their boundaries. Previous studies have concluded that most of Amazonian Brazil's ecological reserves lack adequate management on the ground (Peres and Terborgh 1995, ecological reserves are roughly synonymous with our high protection reserves). Many high protection reserves are only accessible by boat or plane and are located in remote regions. In areas where there is considerable pressure to develop, high protection reserves have sustained their forest cover (but see Gascon *et al.* 2001 for the

effects of edge collapse). Reservo Ducke has protected its forested cover in time series analysis between 1977 and 1999 (Figure 4.3, Bergen and Steininger in Chapter Three).

Within this study's modeling scenario, forested high protection reserves and indigenous lands remain pristine. In instances where high protection reserves occur in areas that would be degraded, for example within 2 km of a paved highway (a high degradation class), the reserve's forests will remain unaffected (maintaining forest cover but with smaller scale edge effects as described by Gascon *et al.* 2001). Since one location can actually consist of a high protection reserve, a moderate protection reserve, and indigenous lands, high protection reserves are given priority over all other reserve systems.

Moderate Protection Reserves

Moderate protection reserves are the combination of national and state forests, national forest reserves, extractive reserves, state extractive forests, sustainable-use forests, sustainable development reserves, environmental protection areas and areas of relevant ecological interest (Appendix C). Appendix C's Table C.2 lists activities permitted and banned under specific reserve systems. As noted above, reserve systems are not necessarily mutually exclusive and one reserve may possibly be part of several different reserve designations.

Moderate protection reserves are classified as lightly degraded since all of the reserves within this class allow for harvesting, logging, and limited agriculture and livestock. Moderate protection reserves will negate degradation associated with other infrastructure development projects. Therefore, moderate protection reserves that exist

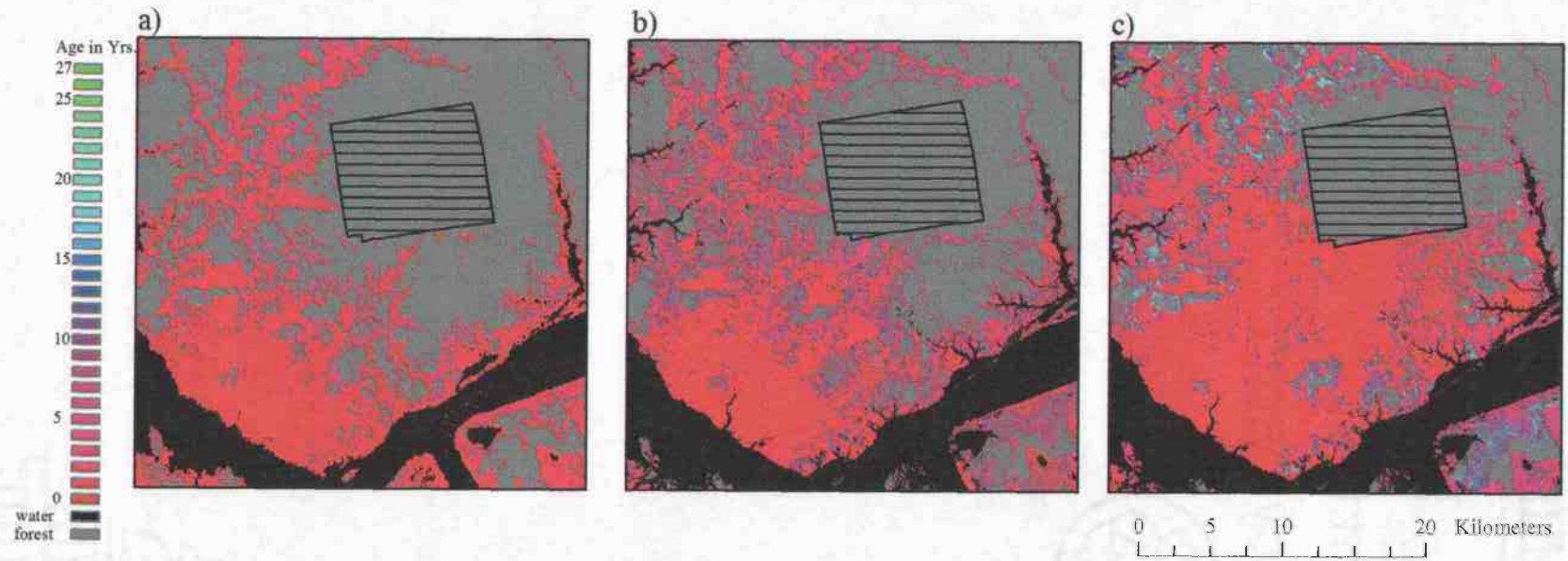


Figure 4.3 Reservo Ducke (bars) in 1977 (a), 1986 (b) and 1999 (c) in classifies MSS and TM imagery depicting secondary forest age, mature forests and water. This high protection reserve is shown to deflect development along its southern and western boundaries. Anthropogenic land alteration is held to a minimum at park boundaries implying that reserves actually prevent development.

within zones that have been degraded by other factors will only be compromised to a lightly degraded class. Moderate Protection reserves are given priority over indigenous lands in this studies' model.

Indigenous Lands

Indigenous lands are comprised of recognized tribal affinities existing within the Brazilian Amazon (Appendix C). Approximately 20% of the entire Brazilian Legal Amazon exists under indigenous land designation, making it the largest single *management system within the entire Amazon Basin*. The PP-G7 nations wished for Brazil to recognize and preserve its indigenous cultures existing within the Brazilian Legal Amazon and contributed \$340 million to this end. At least 375 reserves have been recognized and the federal government is continually updating information on these groups through FUNAI (Brazilian federal institution for the management of indigenous lands and people). Although reserves are delineated, several remain in dubious management and are often ignored by state governments when more economically advantageous land claims assert themselves or are under development and natural resource extraction pressures. For indigenous lands, no title has been transferred to the tribes (with the exception of two).

One important question that concerns indigenous lands and has received scant review is, are indigenous lands supportive of conservation of biological species diversity, ecosystem function, and forest cover? Would these people react to special rules and *license by liquidating their natural resources to acquire short-term economic gain*? The most likely scenario is that indigenous groups will react in relationship to the pressures of

the socio-economic systems in which they are embedded. One trend that is visually apparent is, as surrounding lands succumb to economic pressures to liquidate natural resources, more pressure will be placed on these indigenous societies to follow suit (Figure 4.4). Indirect costs are more likely to play an increasingly important role in the preservation of the tribes as pressures build. Factors such as health care for newly introduced diseases and parasite strains, decreased fishing and hunting capacity and the introduction of more productive agricultural means are more likely to expedite the abandonment of traditional sustainable practices (Tierney 2000, Wallace 2003). In very remote locations, the Brazilian federal government has established an official ‘hands off’ management approach to these societies that exist in a precarious balance. Non-endemic peoples are restricted from indigenous lands and trespassing sometimes results in violent confrontation. Several anthropologists’ view that contact with previously non-contacted tribes starts a process leading to the ultimate collapse of these people’s “universe.” There are several groups on the western Brazilian border with Peru that have yet to be contacted by western civilization (Wallace 2003).

In the model, indigenous lands will remain in a pristine forested condition if no other infrastructural developments encroach upon their boundaries. Indigenous lands will impede deforestation and habitat destruction when infrastructural developments do encroach upon their lands by only allowing deleterious processes to proceed to a lightly degraded condition. Indigenous lands have no priority over other reserve systems that they might be apart of.

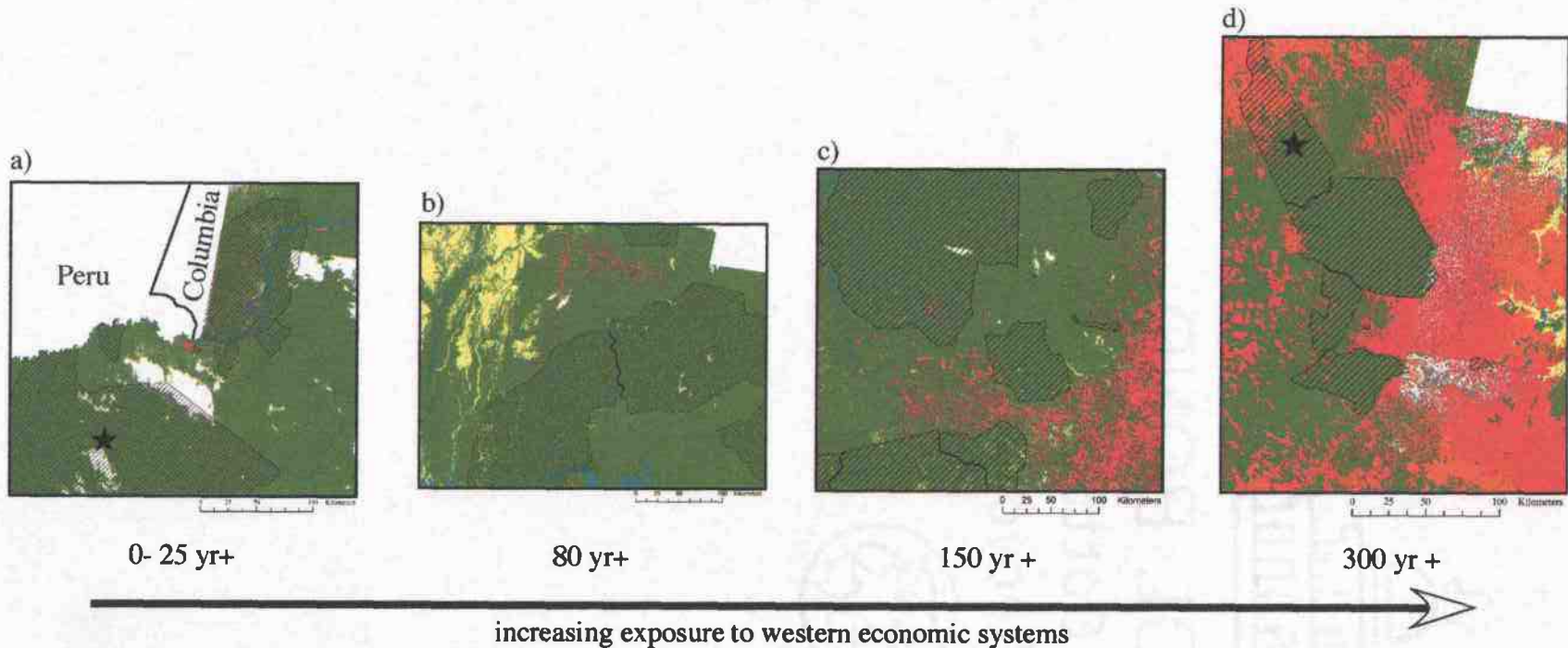


Figure 4.4 A subset of images from the 1992 PathFinder Amazon composite image show deforested areas as red and orange, mature forested areas are green, *cerrado* habitats are tan and clouds with shadows are white and grey. Indigenous lands are delineated by diagonal stripes. Initially, there is not pressure from western economic systems for indigenous groups to exploit their natural resources. Indigenous lands in the far western Brazilian Amazon show negligible deforestation (a). At least four tribal groups occurring in the *Vale do Javari* area (star) have yet to be contacted. North of the city of *Manaus* the territories of the *Waimir-Atroari*, *Trombeta/Mapuera* and *Niamunda/Mapuera* impede linear deforestation caused by roads and hydroelectric project (*Balbina*, 1°30'S 60°01'W, b). Further south on the *Xingu* river, multiple indigenous groups show similar behavior for their approximate 150 years of contact with western economic systems (lower left, *Apyterewa*, *Xicrin do Rio Catete* and *Kayapo*, respectively from top to bottom, 6°45'S, 51°16'W). The figure right (d) shows an indigenous land tenure system SW of *Belem* (*Rio Guuama*, *Alto Turiacu* and *Awa*, 2°41'S, 46°15'W). Notice the indigenous lands are much less deforested than the surrounding areas, in most cases. Certain tribes eventually concede to pressure to liquidate natural resources (star).

Fire Potential

Amazonian forest systems are not adapted to fire. Tree species typically have a bark that is only 2 - 5 mm thick leaving the living tissue extremely vulnerable to small flames (Cochrane *et al.* 1999). Fires that occur within the Amazon basin are typically associated with heavily fragmented and developed agricultural based societies where fires escape from slash and burn agriculture or escape fires from pasture maintenance (Nepstad *et al.* 2001a). Fragmented forest edges have been shown to have differences in their microclimatic conditions (Lovejoy *et al.* 1986) that promote a forests susceptibility to fire (Laurance *et al.* 1997). Nepstad *et al.* (1998) produced a fire potential model of the Brazilian Legal Amazon that operates on the seasonality of rainfall (more extreme in El Nino years) and forest edge. The result of their model is a map of forested areas with high, medium and low fire potential within the Brazilian Legal Amazon (Nepstad *et al.* 1998).

This study considers those forested areas that are classified in the high and medium fire potential categories (Appendix C). Forests that have been recognized as having a high fire potential will be classified as having moderately degraded habitat by the year 2020. Forests having a medium fire potential will be classified as being lightly degraded.

Riparian Deforestation

Riverside deforestation occurs within the Brazilian Amazon but is rarely acknowledged in the literature concerning modern deforestation patterns (but see Laurance *et al.* 2002a). Wood from forests is harvested to supply fuel for steamboat

traffic along the Amazon River and its larger tributaries. Steam powered ferries still exist as a major means of transportation, often being the only means of transportation in the absence of roads or air strips. The cheapest means of transporting goods through the Amazon is by river. Verissimo *et al.* (2001) quantifies that transport of cut timber *via* water is two orders of magnitude cheaper relative to transport by roads or train. It is also worth noting that the majority of timber harvesting areas contain navigable rivers (Appendix C).

We combined data supplied by IBGE for individual states comprising the Amazon and National Oceanic and Atmospheric Administration 1999 Advanced Very High Resolution Radiometer satellite imagery (AVHRR) at 1 km resolution to produce a spatially explicit navigable river coverage. Aquatic systems that are wider than 1 km are considered navigable. Seasonally inundated forests were delineated by IBGE and those occurring adjacent to navigable rivers are considered susceptible to timber harvest (Appendix C, IBGE is the Brazilian federal institution for mapping and statistics). Várzea (inundated forests) are also converted to agriculture because their seasonal inundation elevates nutrient concentrations in the soil (Goulding *et al.* 1995).

Navigable rivers and adjacent inundated forests were selected as having the potential to be associated with deforestation. Riparian deforestation is clearly visible from TM satellite composite images along highly trafficked thoroughfares (Figure 4.5). In this modeling exercise, areas that are within 2 km of a navigable riverbank are classified as highly degraded. Areas 2 to 5 km from the bank are moderately degraded and areas 5 to 10 km are only lightly degraded.

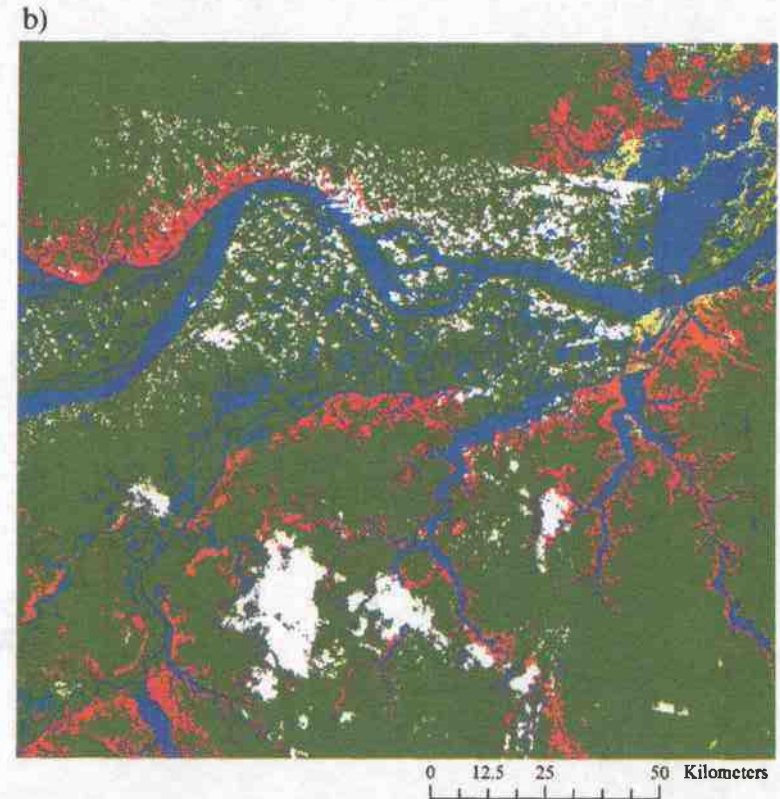
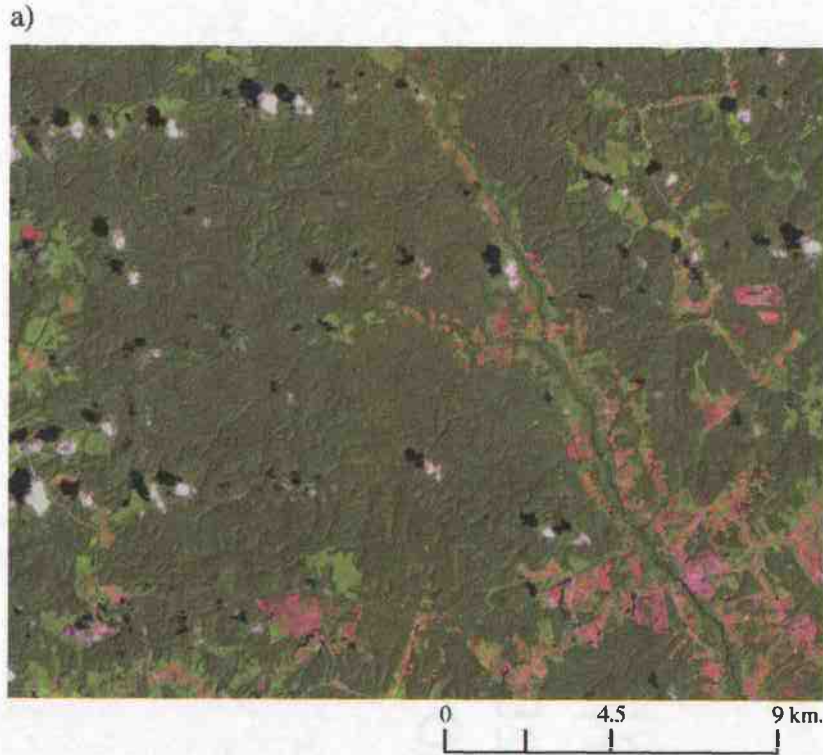


Figure 4.5 Being a major transportation corridor, deforestation can also be spatially auto correlated with navigable river systems. The figure on the left (ETM+ 5,4,3 RGB) is of a small river north of the city of *Manaus* ($2^{\circ}45'S$ $59^{\circ}20'W$) in which river deforestation exhibits the same spatial behavior as that associated with adjacent roads (a). Areas red/pink have been recently cleared. Areas that are lighter green are regrowth forest occurring on previously cleared land. The figure to the right (PathFinder 1992 classified image mosaic) shows deforestation closely associated with navigable river channels along the Amazon River (b) ($2^{\circ}36'S$ $5^{\circ}47'W$, blue = water, red & orange = deforested, white/grey = cloud/ shadow, yellow = savanna and green = forested).

OVERLAY METHODOLOGY AND ANALYSIS

Overlay Methodology

All of the model elements and associated data layers of buffer distances were overlain on a personal computer using GIS software. Data layers were geo-referenced to a Lambert Azimuthal Equal Area projection to minimize distortion and maintain accurate area measurements. The resulting combination of the model elements with degradation class buffer distances resulted in a complex spatially explicit model. In this model, each polygon has 2.305×10^{18} possible combinations based on the specific combinations of *spatial relationships and its composition of modeling elements* (Figure 4.6).

Overlay Analysis

Model elements were evaluated by two methods, one that quantifies the minimum effect and one that quantifies the maximum effect of the model element. The models classify areas within polygons into one of four classes, heavily degraded, moderately degraded, lightly degraded, or pristine. Areas not presently forested were classified as heavily degraded. The analysis conducted is similar to step-wise forward and step-wise backwards multiple linear regression.

In order to evaluate the minimum effect of a model element, the element is omitted from a model containing all other elements. The model that contains all of the model elements is called the Full model. The model quantifying the minimum impact of an element is referred to as the Element Deleted Model (EDM), reflecting the fact that the *element is deleted from the Full model. This modeling scenario represents the maximum*

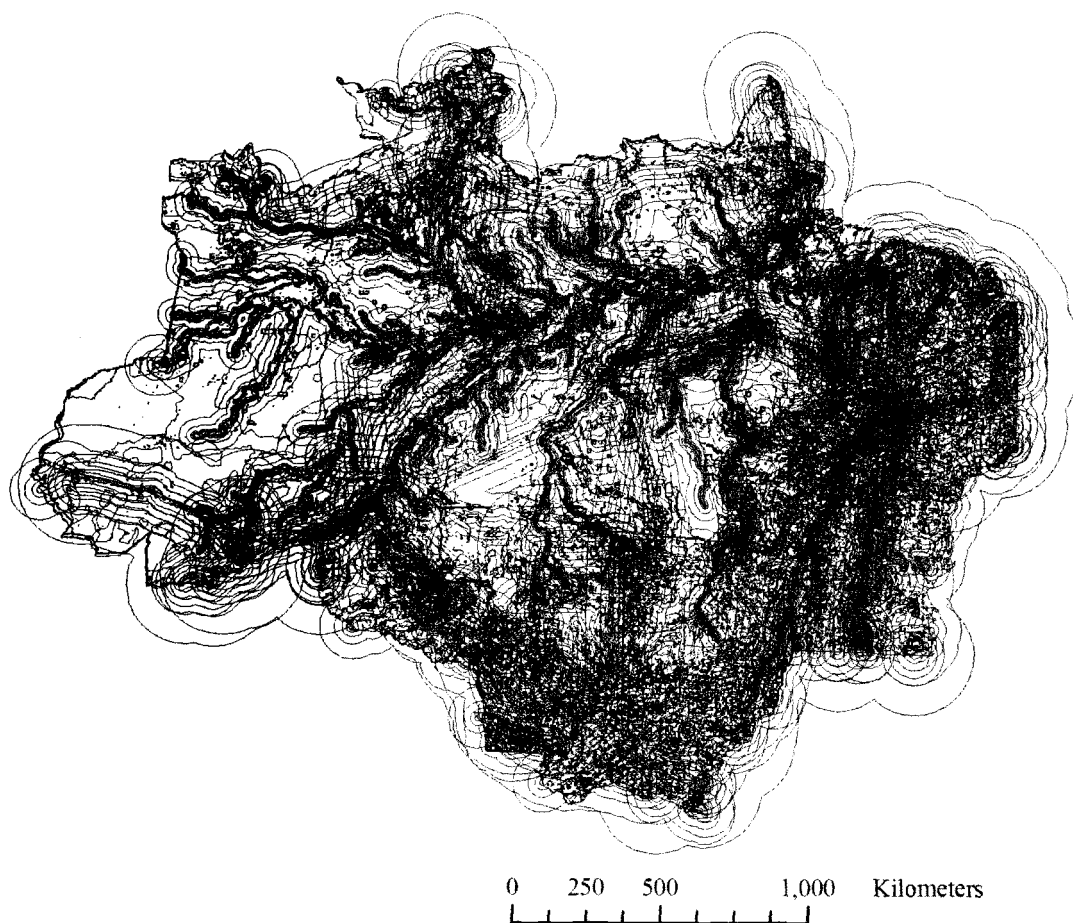


Figure 4.6. The spatial model comprised of all model elements and model element's buffer distances occurring within the Brazilian Legal Amazon. The model contains over 250,000 unique polygons. Each polygon has the potential of 2.305×10^{18} possible unique combinations of model elements.

influence of all other model elements contained within the Full model. Many of the modeling elements occur within the same area, so simply removing just one element will not stop the other elements' effect upon forest. The differences between the Full model and the EDM are recorded across four classes.

In order to quantify the maximum effect of a modeling element, the element is added to a model that only includes present infrastructure (the Null model). This model scenario assumes that all other future development projects will not progress with the exception of the element in question. The difference between the Null model and the *Element Added Model (EAM)* are recorded for all of the four classes.

Confounding the clarity of the analysis, some of the model elements grow in spatial extent in the near future while other model elements are static. This disparity requires further delineation of these temporal differences within the EDM and EAM methods. In order to estimate their contribution to degradation, deforestation, or resource conservation, static elements were also used the EDM and EAM scenarios, as in the cases of fire probability, reserve systems, timber concessions, mining concessions, and riparian deforestation.

Certain model elements were further grouped to quantify the total impact of these combined development projects. Road improvements and new roads, near future and late future dams, timber and mining concessions, high protection, moderate protection and indigenous lands were combined to produce the elements of all road activity, all planned dams, natural resource depletion and total reserve systems, respectively. Besides aiding in the generalization of infrastructural projects, it also improved quantification of the

effects of Brazil's mixed reserve systems where one location can belong to all three reserve classes.

After EDM and the EAM scenarios were completed, the differences for all 17 elements were recorded for their effect over the study area in km². The differences are the minimum and the maximum that the element will impact future forest within the study area within a particular class. The numbers recorded for each class within the EAM and EDM scenarios are also averaged.

Core areas of pristine forest (areas > 10,000 ha.) were examined. Forests of at least this size are necessary to provide a full species component with ecological integrity (Terborgh 1988, Terborgh *et al.* 2001, Laurance *et al.* 2002b). To assess the risk of fragmentation to these remaining large forests tracts, we estimated the effect of model elements on pristine forests that are larger than 10,000 ha.

The resulting element models from the EDM and EAM scenarios served for the analysis of fragmentation within the study area. For this, we combined the lightly degraded and pristine forest classes within each of the element models. Lightly degraded forests are defined by having a small percentage of their forests deforested (< 5%). Degradation of this degree has a nominal effect on forest cover and fragmentation processes since most of the supporting forest matrix is sustained. The element models then dissolved pristine and lightly degraded classes in a spatially explicit manner to produce accurate estimates of the size class distributions of non-fragmented forests in the year 2020.

RESULTS

The results from comparison of the EAM (maximum effect) and the EDM (minimum effect) are tabulated by their classes of land alteration for the elements of all road activities, road improvements, new road construction, all hydroelectric projects planned for development, hydroelectric projects slated for development before 2010, river channelization projects, new railroads, new utility corridors, natural exploitation areas, timber concessions, mining, reserves systems, federal ecological reserves, state sustainability reserves, indigenous lands, fire, and riparian deforestation (Table 4.1). The results listed are for elements that accelerate deforestation and those that inhibit it. Table 4.1 lists how each of these factors contributes to the spatial area of each class. Positive numbers mean that the element contributed more area to the habitat class. Negative numbers mean that the model element decreased the area of the habitat class. For example, all road activity increases the distribution of heavily degraded habitats by 84,442 km², moderately degraded habitats by 6,224 km², lightly degraded habitats by 13,803 km² and decreases pristine habitats by 104,468 km² within the EAM scenario. The average taken from the EAM and EDM models are reported (Table 4.1). The percent of each elements' contribution to each habitat class is calculated for EAM and EDM methods with the mean (Table 4.2). Core areas of pristine forest are delineated with EAM and EDM methods with the average of the two reported (Figure 4.7). Areas of non-fragmented forests are quantified via EAM and EDM scenarios with their average (Figure 4.8).

Table 4.1 Calculated area effects of model elements from the Element Added Model, and Element Deleted Model and averaged results scenarios for four different classes of land alteration occurring within the Brazilian Amazon by the year 2020.

	Element Added Model (Sq Km)				Element Deleted Model (Sq Km)				Average of Models (Sq Km)			
	Heavily Degraded	Mod. Degraded	Lightly Degraded	Pristine	Heavily Degraded	Mod. Degraded	Lightly Degraded	Pristine	Heavily Degraded	Mod. Degraded	Lightly Degraded	Pristine
All Road Activity	84,442	6,224	13,803	-104,468	80,413	1,668	13,245	-95,325	82,428	3,946	13,524	-99,897
Road Improvements	73,829	-1,620	-15,453	-56,756	69,585	-6,191	-15,818	-47,576	71,707	-3,905	-15,635	-52,166
New Road Construction	10,828	7,859	29,063	-47,749	10,613	7,844	29,256	-47,712	10,721	7,852	29,159	-47,731
All Planned Dams	18,807	-3,948	4,821	-19,679	15,420	-4,370	4,241	-15,290	17,113	-4,159	4,531	-17,485
Near Future Dams	14,908	-4,437	2,282	-12,753	11,521	-4,859	1,702	-8,364	13,214	-4,648	1,992	-10,558
Channelization Projects	9,885	802	-4,399	-6,287	9,725	346	-3,785	-6,287	9,805	574	-4,092	-6,287
New Railroads	13,179	7,370	-2,334	-18,215	11,493	3,208	-3,194	-11,507	12,336	5,289	-2,764	-14,861
New Utility Corridors	17,262	753	4,661	-22,676	15,412	1,054	4,668	-21,133	16,337	904	4,664	-21,905
Without Nat. Resource Depletion	0	164,395	35,834	-200,228	0	141,820	12,894	-154,713	0	153,107	24,364	-177,470
Without Timber Concessions	0	164,395	-6,147	-158,247	0	141,820	-23,081	-118,738	0	153,107	-14,614	-138,493
Without Mining Areas	0	0	40,855	-40,855	0	0	34,849	-34,849	0	0	37,852	-37,852
Without Reserve Systems	151,777	277,269	-428,981	-67	203,572	295,684	-479,571	-19,687	177,674	286,477	-454,276	-9,877
Without High Protect. Reserves	31,639	26,677	56,854	-115,171	33,736	29,178	64,777	-127,691	32,687	27,928	60,816	-121,431
Without Mod. Protect. Reserves	46,882	81,620	-234,847	106,344	57,048	81,635	-237,999	99,315	51,965	81,627	-236,423	102,830
Without Indigenous Lands	66,476	149,633	-216,109	0	101,481	163,143	-264,625	0	83,979	156,388	-240,367	0
Without Fire	0	30,136	29,820	-59,959	0	21,535	24,139	-45,677	0	25,836	26,979	-52,818
Without Riverine Deforestation	143,058	26,577	80,744	-250,376	125,683	13,944	56,536	-196,162	134,370	20,260	68,640	-223,269

Table 4.2 Percentage of total land alteration due to model elements from the Element Added Model, Element Deleted Model and averaged results scenarios for four different classes of land alteration occurring within the Brazilian Amazon by the year 2020.

	Element Added Model (%)				Element Deleted Model (%)				Average of Models (%)			
	Heavily Degraded	Mod. Degraded	Lightly Degraded	Pristine	Heavily Degraded	Mod. Degraded	Lightly Degraded	Pristine	Heavily Degraded	Mod. Degraded	Lightly Degraded	Pristine
All Road Activity	19.56	1.25	1.98	-11.58	17.85	0.36	1.80	-12.35	18.71	0.81	1.89	-11.96
Road Improvements	17.10	-0.32	-2.13	-6.29	15.46	-1.31	-2.06	-6.16	16.28	-0.81	-2.10	-6.23
New Road Construction	2.51	1.57	4.00	-5.29	2.36	1.65	3.81	-6.18	2.43	1.61	3.91	-5.74
All Planned Dams	4.35	-0.79	0.66	-2.18	3.43	-0.92	0.55	-1.98	3.89	-0.85	0.61	-2.08
Near Future Dams	3.48	-0.88	0.32	-1.42	2.58	-1.02	0.22	-1.09	3.03	-0.95	0.27	-1.26
Channelization Projects	2.29	0.16	-0.61	-0.70	2.16	0.07	-0.49	-0.81	2.22	0.12	-0.55	-0.76
New Railroads	3.05	1.47	-0.32	-2.02	2.55	0.68	-0.42	-1.49	2.80	1.07	-0.37	-1.75
New Utility Corridors	4.00	0.15	0.64	-2.51	3.42	0.22	0.61	-2.74	3.71	0.19	0.63	-2.63
Without Nat. Resource Depletion	0.00	32.79	5.01	-22.16	0.00	29.90	1.79	-20.01	0.00	31.35	3.40	-21.09
Without Timber Concessions	0.00	32.79	-0.85	-17.54	0.00	29.90	-3.01	-15.38	0.00	31.35	-1.93	-16.46
Without Mining Areas	0.00	0.00	5.63	-4.53	0.00	0.00	4.54	-4.51	0.00	0.00	5.09	-4.52
Without Reserve Systems	-34.60	-53.25	66.27	0.01	-44.11	-59.61	70.61	3.49	-39.36	-56.43	68.44	1.75
Without High Protect. Reserves	-7.33	-5.32	-7.83	12.76	-7.49	-6.15	-8.45	16.54	-7.41	-5.74	-8.14	14.65
Without Mod. Protect. Reserves	-10.86	-16.28	32.34	-11.79	-12.67	-17.21	31.03	-12.87	-11.76	-16.75	31.69	-12.33
Without Indigenous Areas	-15.39	-29.84	29.76	0.00	-22.54	-34.40	34.50	0.00	-18.97	-32.12	32.13	0.00
Without Fire	0.00	6.01	4.11	-6.65	0.00	4.54	3.15	-5.92	0.00	5.28	3.63	-6.28
Without Riverine Deforestation	33.13	5.30	11.12	-27.75	27.92	2.94	7.37	-25.41	30.52	4.12	9.25	-26.58

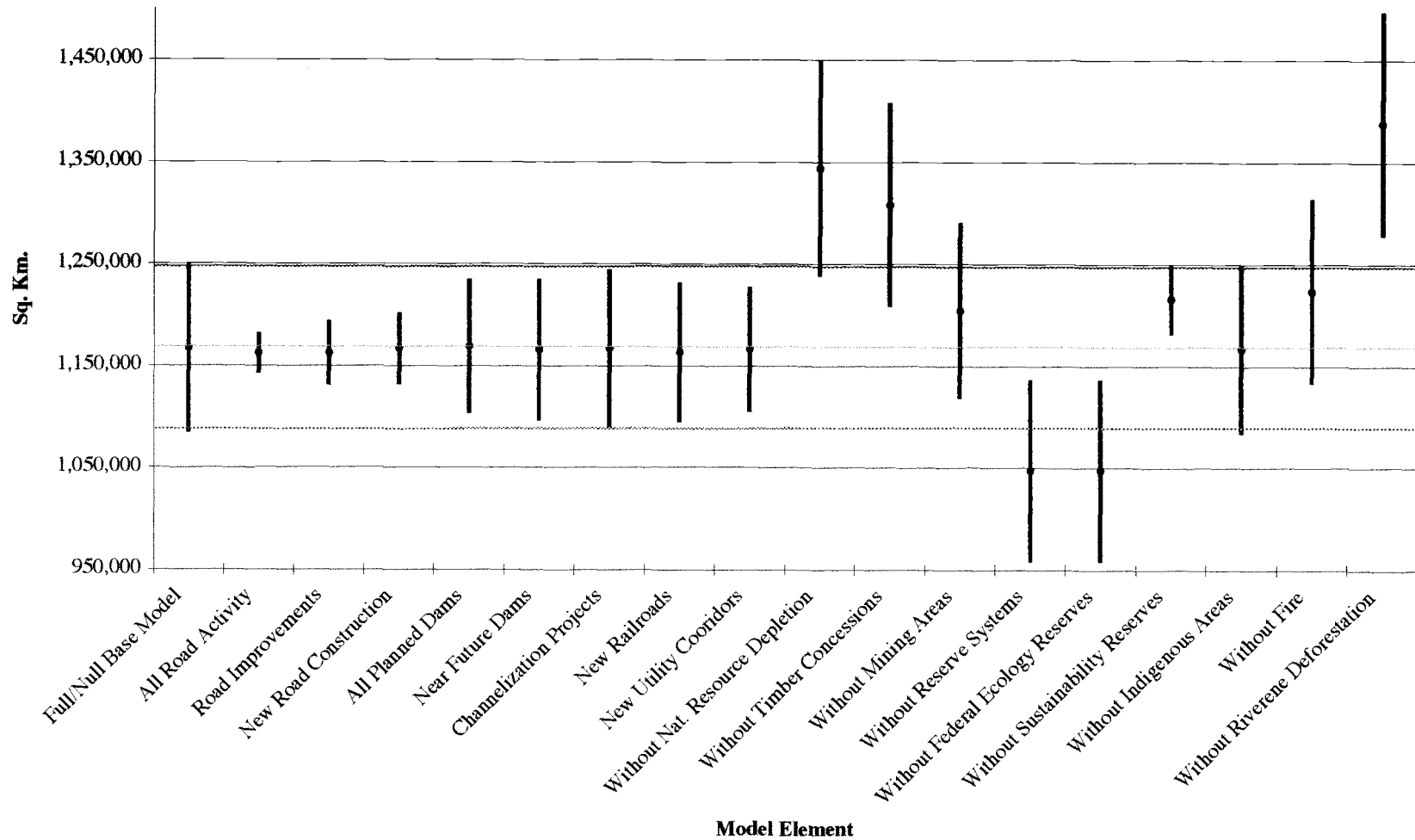


Figure 4.7. The Area of Pristine core forests (> 10,000 ha.) under the element added (upper bar), element subtracted (lower bar) and averaged modeling scenarios (dot). The base NULL model used for the element added model (lower dotted line) and the FULL model for the element deleted model (upper dotted line) are included for reference with their average (red line).

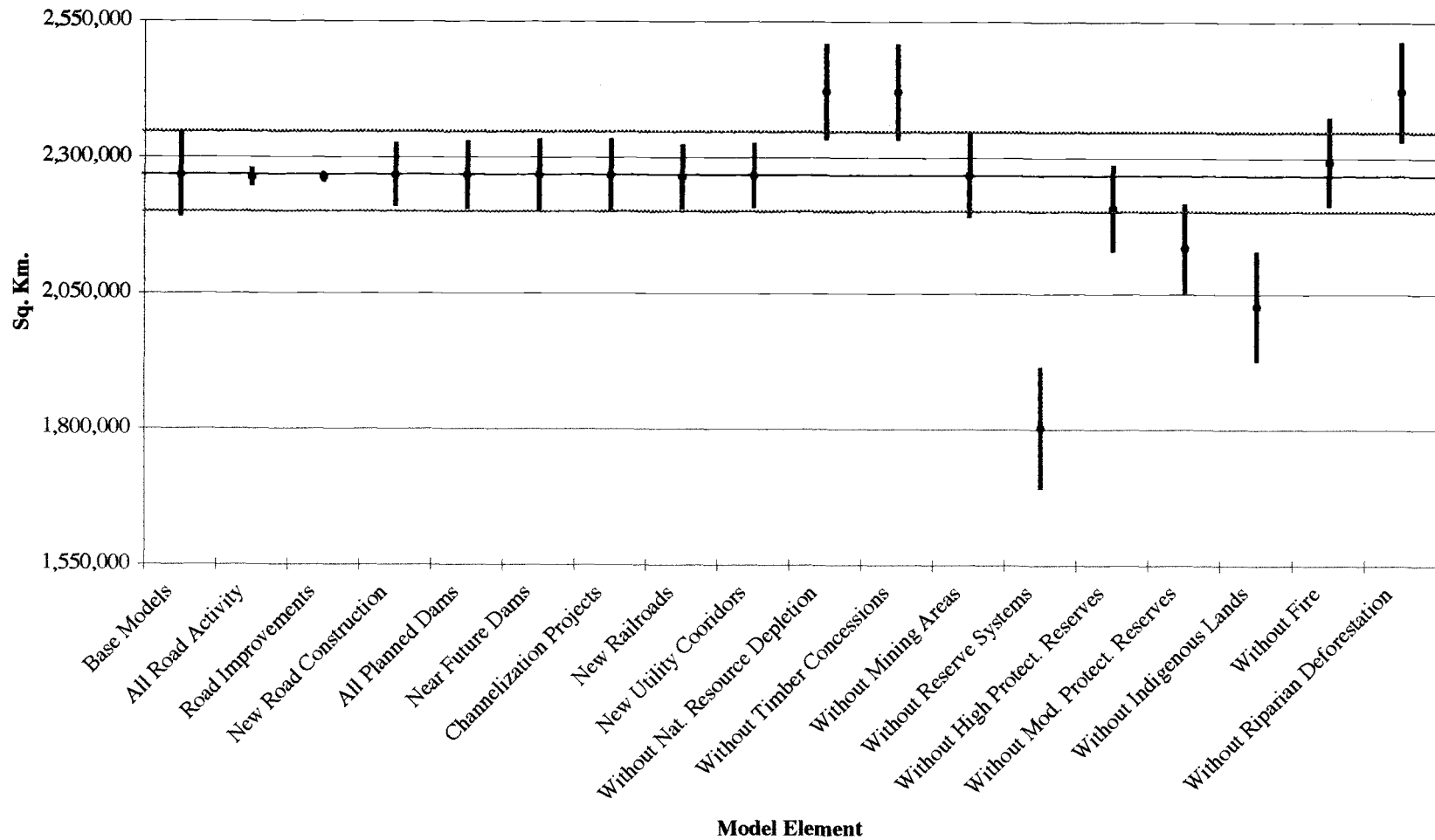


Figure 4.8 Total area of non-fragmented forests as quantified by Element Added and Element Deleted Model scenarios (upper and lower bars) with the average (dot). The upper and lower dashed line represent the Null and Full models with a red line of their average.

ELEMENT RESULTS

The results for each of the elements contained in the EDM and EAM scenarios are listed below. The results of the effects on core pristine forests and fragmented forests are presented for paired EDM and EAM and the Full and Null models.

All Road Activity

All road development in the modeling scenarios increases the amount of heavy and moderate altered habitats by an average of 86,374 km² with over 95% being in the *heavily modified class* (Table 4.1). *An additional 13,524 km² is affected in the lightly degraded class.* Road activities further decrease almost 100,000 km² of pristine forested habitats. Over 99% of the land altered is derived from pristine forested habitats.

Road Improvements

Road improvements increase the spatial distribution of heavily modified habitats by 71,707 km². Moderate and lightly modified habitats decrease because the area reduced, 19,540 km², is shifted to the heavily modified class. An average of 52,166 km² of pristine forest is altered by road improvements. Seventy-two percent of the changes are from the loss of pristine forests.

New Road Construction

New roads increase heavy and moderately altered habitats by 18,573 km² with *58% existing in the heavily degraded class.* *Lightly modified habitats also increase by 29,159 km².* The majority of the lands degraded are of pristine forest origin.

All Hydroelectric Dams

All hydrologic projects planned for by the year 2020 within the Brazilian Legal Amazon increase heavily modified habitats by 17,113 km². Approximately, 4,159 km² of lands heavily degraded are derived from existing habitats that are moderately degraded forests. Another 4,531 km² is added to the lightly modified class as pristine forest are decreased by 17,485 km². Seventy-six percent of all lands degraded were of pristine forest origin.

Hydroelectric Dams To Occur Before 2010

Dams to be built in the near future (with some presently under construction) further heavily degrade 13,214 km² of forested habitats, 35.2% of which are previously classified as moderately degraded lands. Further, 1,992 km² is lightly degraded from pristine from condition. A total loss of 10,558 km² of pristine forests is estimated. Of the total land degradation by near future hydroelectric projects, 65% are from pristine forests.

River Channelization

River channelization accounts for an increase of 10,379 km² of heavily degraded land. Approximately 40% of the heavy and moderately degraded lands come from lands that are previously classified as lightly degraded. Channelization accounts for the degradation of 6,287 km² of pristine forest habitat.

New Railroads

Railroad construction account for an increase of 17,625 km² in heavy and moderately degraded habitats with 16% coming from lightly degraded forests and the remainder from pristine forests. Railroad construction degrades 14,861 km² of pristine forest habitat.

New Utility Corridors

Utility corridors account for an increase of 17,241 km² of heavy and moderately degraded habitats with 95% occurring in the heavily degraded class. Another 4,664 km² of lightly degraded forest are created from 21,905 km² of pristine forest that is degraded by utility corridor construction.

Natural Resource Depletion

When presently occurring natural resource extractive activities are not included in the EAM and EDM scenarios, no increase in heavily modified habitats are observed. Moderate and lightly degraded classes decrease by 164,395 and 35,834 km², respectively, while a total of 200,228 km² is conserved as pristine forest in the absence of concessions. Natural resource extraction is the second largest cause of forest fragmentation (Figure 4.8) and pristine core forest reduction (Figure 4.7).

Timber Concessions

Timber concessions have no effect on the spatial distribution of heavily modified habitats. Moderately disrupted habitats decline by 153,107 km². Whereas lightly

modified habitats increase by 14,614 km². An additional 138,493 km² are added to the pristine forest class. Timber concessions comprise all of the effects for forest fragmentation in natural resource depletion, making it the second largest forest fragmentation promoter and a substantial consumer of core forests.

Mining

Mining, concessionary and wildcat, has no influence on the distribution of heavy or moderately disturbed habitats. Without mining lightly disturbed habitats decreased by 40,855 km² that are of pristine forest origin.

All Reserve Systems

The absence of reserve system led to dramatic increases in heavily modified habitats, 177,674 km². Moderately disturbed habitats increase further by 286,477 km². Counter intuitively, lightly disturbed habitats decrease by 454,276 km² since sustainability reserves create lightly disturbed habitat. Pristine forests lost 9,877 km². Collectively, reserve systems are shown to be the greatest inhibitor of forest fragmentation and provide the best means to preserve core forest systems.

High Protection Reserves

Removal of high protection reserves led to the increase of 32,687 km² of heavily modified habitats. Moderate and lightly degraded habitats would also increase by 27,928 km² and 60,816 km², respectively. Pristine forests are substantially decreased, 121,432

km². High protection reserves are shown to inhibit forest fragmentation marginally but protect core forests better than any other singular model element.

Moderate Protection Reserves

Removal of sustainability reserves led to an increase of heavy and moderately disturbed habitats, by 51,965 and 81,627 km², respectively. Lightly disturbed habitats decrease by 226,423 km². Pristine forests increase their distribution by 102,830 km². Moderate protection reserves increase forest fragmentation, and these reserves contribute to the alteration of core pristine forests.

Indigenous Lands

In the absence of indigenous lands, heavy and moderately modified classes increase by 240,367 km², with 65% occurring in the moderately disturbed condition. Nearly all of this land is derived from previously lightly degraded habitat. Reserves of indigenous peoples exerted no influence on the distribution of pristine forest habitats since this model element only acts upon degraded habitat classes within its boundaries. Indigenous lands are found to have the greatest effect on preventing forest fragmentation of the three reserve systems investigated. Indigenous lands have no effect upon core forest distribution since the model element's behavior in the EDM and EAM scenarios only change heavily and moderately degraded habitats to lightly degraded.

Fire Potential

In the absence of fire events, heavily disturbed habitats did not change their spatial distributions. Moderate and lightly degraded habitats decrease by 25,836 and 26,979 km², respectively. Pristine forests have a 52,818-km² increase. Forest fires accelerate forest fragmentation and consume core forests.

Riverside Deforestation

Without riverside deforestation heavy, moderate and lightly degraded habitats would decrease by a total of 223,270 km². This land would be pristine forest if river deforestation had not occurred. Riparian deforestation is the largest threat to forest fragmentation and core forest distribution within the Brazilian Legal Amazon.

DISCUSSION

MODEL METHODOLOGY

Because we examined individual element models, our interpretation of deforestation is more complex, and we believe it is more complete than traditional forest/non-forest models. The element models follow where development occurs and the magnitude of the degradation. These element models allow us to determine what type of habitat is altered and provide an estimate of a given element's spatial effects on forested habitat. By comparing the amounts of each habitat class with the distributions of these *classes in the Null and Full models*, we are able to increase our understanding of the magnitude of forest degradation. Within this exercise, habitat classes degrade from pristine to lightly degraded to moderately degraded to heavily degraded. Thus, the EDM and EAM methods produce valuable results in quantifying how different infrastructural developments occurring in the Brazilian Amazon will contribute to deforestation, habitat degradation, and the conservation of pristine habitats by the year 2020.

We find our modeling approach relatively robust in that the EDM and EAM methodologies produced similar results in the ranking of each of the model elements relative to the class of degradation (Table 4.2). However, there are large differences in the amounts that each element affects the different classes (Table 4.1). Using the EAM methodology, future planned developments result in a higher ranked effect of a given modeling element relative to EDM methods. We found similar results for extractive concessions present within Brazilian Amazonia. The reserve elements within the EAM *methodology are less than what is quantified for the EDM methodology*. The relation of development and parks within a more complete "build-out" scenario, *i.e.* the EDM

scenarios, may explain this seemingly counter intuitive result. Because under the EDM methodology all other developments proceed except the one element under investigation, the number of future projects encountering reserve systems is much greater, thus the reserve systems have greater influence relative to the results from EAM scenarios. This results from our intent to quantify the minimum effect of an element, so we consider this a conservative approach.

The next consideration in interpreting the EDM and EAM methodologies is that the numbers reported are differences between either the Null or Full models and a *particular element model*. *The differences are reported for all habitat degradation classes*. Under the EAM scenario, all present infrastructure builds out to the spatial parameters specified. By measuring the difference between the Null model and the EAM in question, we can estimate how much land will be influenced by a particular model element. Under the EDM scenario, we are measuring the differences between the Full model and a full model where the element has been deleted. The difference between the Full model and the EDM model is the minimum amount of influence that the element has across the four degradation classes. In these scenarios, infrastructural development projects that occur in remote areas report a greater influence relative to areas that are less remote and generally have more development by other sources. Since the model identifies infrastructure that is already on the ground, the results are less dramatic than what has been presented in previous studies (Nepstad *et al.* 2001a). These studies have measured the amount of absolute change where planned development and present *development infrastructure have not been distinguished*.

AVANÇA BRASIL INFRASTRUCTURE PROJECTS

Our model provides estimates suggesting that habitat degradation from *Avança Brasil* will be extensive. *Avança Brasil* is the combination of the elements of road improvement, near future dams, river channelization projects, new railroads and new utility corridor construction. On average, all of the *Avança Brasil* projects account for a 123,399 km² increase in heavily degraded habitats, reduce moderately degraded habitats by 1,786 km², reduce lightly degraded habitats by 15,835 km² and degrade 105,777 km² of pristine mature forest habitats. Only 14% the heavily degraded areas are from areas designated as moderately or lightly degraded, suggesting that the majority of *Avança Brasil* projects will impact areas that are previously inaccessible or undisturbed. This analysis suggests that because heavily degraded areas represent 50% forest cover, moderately degraded habitats have 85% forest cover and lightly degraded forests still retain 95% forest cover, *Avança Brasil* will lead to a minimum of 54,470 km² of deforestation beyond that which is projected from current infrastructure within Brazilian Amazonia. With a conservative estimate of Amazonia forests holding 140 Mg Carbon ha⁻¹, the amount of carbon added to the atmosphere will increase previous estimates of 0.2 Pg Carbon yr⁻¹ by 19% for the next twenty years (Houghton *et al.* 1999, 1 Pg is 10¹⁵ g). These calculations do not include late series dams, new road developments or El Nino fires that occur through out the study area.

The projects contained within *Avança Brasil* all contribute to deforestation, habitat fragmentation, sources of atmospheric carbon, and deforestation. Road improvements will produce the majority of the heavily degraded habitats (58%) and be responsible for almost half of the losses incurred in pristine forest habitats (49.3%). This

is consistent with other findings examining road improvements (Laurance *et al.* 2002a, Nepstad *et al.* 2001a). Utility corridor construction will be responsible for 13% of the heavily degraded habitats and 21% of the decreases in primary forest. Hydroelectric projects are responsible for 11% of the heavily degraded habitats and 10% of the pristine forest loss. New railroads will represent 10% of the increase in heavily degraded forested habitat in *Avana Brasil* but are responsible for 14% of pristine forest loss. River channelization projects represent 8% of heavily degraded habitats and are responsible for a decrease of 6% of the pristine forest.

The effects of Avana Brasil upon pristine forests of greater than 10,000 ha are limited relative to the variance associated with the factors of natural resource depletion, fire, reserve systems, and riparian deforestation. Core pristine forests modulate by some 80,000 km² about a mean of 1,167,000 km² (Figure 4.7). The largest disparities in variance occur with the near future dams, channelization projects, and railroads. These differences are because of the relatively small influence of individual projects types relative to the larger size of all Avana Brasil projects (as in the case of EDM scenarios) or the small area of added degradation relative to present infrastructure (as in the case of EAM scenarios). Elements with smaller variance show that the projects will compromise a more definitive area because of the project's long linear spatial arrangement, as is the case with utility corridors and road upgrades. Road improvements show less variability because they are a major factor in both EDM and EAM methodologies. Road improvements have a large effect on decreasing the core pristine forest area when added to the Null model and likewise have a large effect on core forests when they are subtracted from the Full model.

The EDM and EAM scenarios for fragmentation analysis are very similar to those reported for core pristine forests. Road improvements, new railroads, utility corridors, near future dams and riverside deforestation are listed in order of greatest to least effect on large-scale fragmentation processes (Table 4.1). On average, the *Avança Brasil* projects will leave 2,268,327 km² (\pm 4,408 km²) of non-fragmented forest, less than 50% of the Brazilian Legal Amazon. The influence of utility corridors and railroads could be greatly decreased as well as be more economical if the projects were concentrated along existing roads. By concentrating the transportation and utility corridors in one location, *surrounding development might proceed at an increased magnitude but paths of* penetration into pristine forests would be minimized. Not only would this help restrict access by hunters and illegal timber harvest, it would also decrease the area of potential ignition sources. Forest fires from uncontrolled ignition sources can cause catastrophic forest fires that proceed well beyond their initial sources affecting thousands of km² (Nepstad *et al.* 2001a).

The *Cuaba-Rio Branco, Cuiba-Manaus* gas pipeline is a perfect example of how western Amazonia frontier expansion can be significantly restricted but still yield comparable service at a lower economic cost by using existing infrastructure. By reconfiguring this project to use BR-319 with a 'T' junction routing from BR-319 to *Cuiba*, 16,344 km² of pristine forest could be saved. Clean development mechanisms identified in the Kyoto Protocol, and accepted by European Union carbon market trading, can serve as a financial vehicle for re-engineering, since changing a proposed *infrastructural development project will save this forest carbon from entering the* atmosphere.

INFRASTRUCTURE PROJECTS BEYOND *AVANÇA BRASIL*

Late schedule Dams and new road construction (primarily the construction of the North Amazon Highway) are not as harmful as their respective *Avança Brasil* counterparts. Late stage hydroelectric projects are one third as destructive as near future dams but still will compromise an average of 7,500 km² of pristine forest. The late schedule dams did not consume a high proportion of habitat that was already degraded, indicating that these areas are penetrating into previously untouched forest. The North Amazon Highway's creation of 10,000 km² of heavily degraded habitat is one seventh of road improvements under *Avança Brasil* but will increase moderately and lightly disturbed habitats far in excess of road improvements. This is because the Northern Amazon Highway will be created from pristine forest previously too remote for extractive economic forces to be profitable. Not surprisingly, if constructed, this one highway would almost consume as much pristine forest as the entire road improvement projects under *Avança Brasil*. Its construction would also imperil most of the Guyana shield's forest structure by fragmenting one of the largest non-fragmented forests within the Amazon basin. The Guyana shield is recognized as a region of endemic mammalian diversity (Patton *et al.* 2002), avian diversity (Haeffer 1987), and tree species diversity (Prance 1973, 1982) and has not been as impoverished as other geologic forest formations of Amazonia, such as the more southern Brazilian shield.

We estimate that riverside deforestation is the largest single contributor to forest degradation and pristine forest consumption. We believe that the model exaggerates the consumption of riparian forests because the model does not account for the distance to

resource markets, *i.e.* the central place theorem (deforestation is equal to or proportional to population density times cost of transportation to economic markets). Without the development of such markets it is likely that the model over estimates the deforestation that will occur by 2020. Nonetheless, riverside deforestation may be an under-appreciated cause of deforestation. Recent scientific analyses often ignore the effects of earlier steam transit, and ignore rivers as the central means by which the majority of peoples in western Amazonia transport themselves, both indigenous and non-indigenous. Several additional factors may promote riparian deforestation. First, by two orders of magnitude, *rivers are the most economically efficient means to transport lumber to markets* (Verissimo *et al.* 2001). Obviously, the trees closest to the river will be harvested first. Second, all of the timber concessions within Brazilian Amazonia have large navigable rivers running through them. The *Tocantins* river system, where 85% of seasonally inundated *várzea* forest has been felled, is a good example of the extent of riverside deforestation in areas where there are economic markets in close proximity (Bergen *et al.* 2003). Large Amazon basin rivers are one of the primary features responsible for isolating bio-regions within the Amazon and under the Pleistocene refugia hypothesis, bioregions are usually delineated by river boundaries (Haeffer 1969, 1987, Prance 1973, 1982, Brown and Ab'Sáber 1979, Brown 1982, Salo 1987, Vanzolini and Williams 1970, Meggars 1975, Hall and Harvey 2002, Lynch 1979, Patton *et al.* 2000). Therefore, rivers might act as part of the natural isolating factors within the Amazon basin that promote allopatric speciation.

Fire did not influence as much land as had been previously reported (Nepstad *et al.* 2001a) despite the fact that we used some of their published spatial data on fire

potential (Nepstad *et al.* 1988). Fire is not synergistic with other model elements, whereas in other models fire was associated with improved and unimproved roads as an ignition source. This modeling scenario quantified fire events that are associated with roads as part of the degradation that would occur in proximity to roads. Our model did consider areas that are prone to fire because the fire prone areas are either highly fragmented, have a long seasonal dry period, or both. With these technical but important distinctions mentioned, fires are responsible for degrading 45,677 km² of pristine forest into moderately and lightly degraded forest under the EAM scenario. This area could be elevated to almost 60,000 km² if all infrastructural developments are to take place. These amounts are an order of magnitude less than what has been described elsewhere (Nepstad *et al.* 2001a). Fires are quantified to have a limited effect on the distribution of fragmented forests. This is consistent with other studies examining fire within the Amazon basin that have concluded that forest fragmentation predates fire susceptibility (Nepstad *et al.* 1998, Cochrane *et al.* 1999, Nepstad *et al.* 2001a).

NATURAL RESOURCE EXTRACTION

With pressure to establish environmental practices that inhibit clear-cutting of forests, Brazil has regulations that limit the amount of timber harvest that can take place within a harvesting area. Commonly, timber is harvested in a patchwork of selective cut forestry that is unidentifiable from TM satellite imagery but is clearly apparent on the ground (Nepstad *et al.* 1999). Areas platted for timber extraction are classified by the model as having no effect on the distribution of heavily degraded habitats. Timber areas will increase moderately degraded habitats by 153,107 km², making them the single

largest contributor to moderately degraded habitats. Timber extraction is the second largest factor depleting pristine forest composition as well as the primary factor contributing to large-scale forest fragmentation. These less extractive measures still have long-term consequences to forest health. Research has shown that displacement of canopy trees opens the forest to increased solar radiation. This increase in solar radiation has the effect of increasing forest temperatures, decreases relative humidity, and also serves as a locus for continued tree falls from wind sheer (Lovejoy *et al.* 1986, Laurance 2000). This drying of the forest creates conditions that are much more susceptible to ground fires. Small ground fires can remove as much as 70% of the total above-ground living biomass within five years post ignition (Cochrane *et al.* 1999, Nepstad *et al.* 2001a). Timber extraction accounted for the majority of the degradation caused by all the types of natural resource extraction.

Mineral extraction affects 37,000 km² of pristine forest by compromising them into the lightly degraded category. Mining is not seen to contribute to the further fragmentation of forest, partly because of the models classification of mining areas as lightly degraded. Mining in this model is mainly concerned with natural gas, iron ore, bauxite, gold, uranium, manganese and copper. The spatial behavior of these types of extraction are viewed as less destructive when compared to petroleum extraction in the Ecuadorian Amazon where pipeline and oil pumping stations dominate the landscape and have elevated mercury concentrations in the soil and waterways. However, considering that gold mining typically involves the use of mercury, mercury poisoning and *sequestration within the food chain is still a major concern associated with wildcat gold extraction*, and may be an important issue since regulations are rarely enforced.

The combined effects of timber and mining processes have dramatic influence on pristine core forest distribution and non-fragmented forest spatial distribution. This means that these activities remain the largest threat to forest health within Amazonia based on the spatial data provided by federal Brazilian sources. With the increasing pressure and focus upon contemporary infrastructural development (Laurance *et al.* 2001a, Nepstad *et al.* 2001a), natural resource extraction still occurs in a greater area than all of the areas affected by *Avana Brasil* combined. Although modern forestry practices do limit the total take of forest canopy cover, timber harvesting affects forests well beyond the lines delineated by more traditional forest/ non-forest metrics derived by TM and Enhanced Thematic Mapper satellites. Nepstad *et al.* (1999) reported that an area classified as 15% deforested had 90% of its forests altered by the combined effects of tree harvesting and ground fire. Logging practices have been shown to have severe impacts on the endemic fauna that is hunted in these areas without regulation (Robinson *et al.* 1999, Laurance 2001, Fa *et al.* 2002, Peres and Lake 2003).

RESERVE SYSTEMS WITHIN THE BRAZILIAN LEGAL AMAZON

Reserve systems that are presently delineated within the Brazilian Legal Amazon have the largest influence upon the rates of deforestation, the placement of deforestation, the spatial distribution of pristine core forests, and the distribution of non-fragmented forests within the Brazilian Amazon in the year 2020. Reserves are presently responsible for deferring heavily and moderately degraded forest habitats by 151,777 and 277,269 km² (Table 4.1). With the inclusion of *Avana Brasil* and other projects contained within the Full model these amounts will increase to 203,572 and 295,684 km² for heavily and

moderately degraded habitats. The combined reserve systems are shown to have the most influence of all of the elements investigated for heavily, moderately and lightly degraded forested habitats. Reserves are shown to have the most impact on the spatial distribution of pristine core forests and non-fragmented forest of all the elements studied (Figures 4.7 and 4.8). At present, all reserve systems combined comprise just less than 30% of the Brazilian Legal Amazon.

Reserves systems are categorized by their distinct management structure (Appendix C Tables C.1 and C.2). High protection reserves comprise all of the area *classified as pristine within the combined reserve systems (Figure 4.7). Because there is* less area in high protection reserves relative to moderate protection reserves and indigenous lands, high protection reserves contribute less than other reserves for the perseverance of non-fragmented forests.

Moderate protection reserves are extractive in nature and will be lightly degraded within their boundaries. Without these reserves the area of pristine core forest would increase. However, these reserves are shown to be the second most important reserve management system to combat forest fragmentation, and prevent the further degradation of heavily and moderately degraded forests.

Extractive reserves, such as the moderate reserves described above, stem from the philosophy that in order to preserve nature it might be necessary to harvest it (Verismo *et al.* 2002). One problem with this view is that most of these reserves have unpaved roads and skid-trails to and from the sites of where the extraction is taking place. Usually these *extraction sites are few in number and spaced far apart (usually of the order of several km).* These roads provide a path for further degradation via hunting, and further

penetration of other extractive resources in the future. By trying to surround high protection reserves with sustainable moderate protection reserve, infrastructure that could ultimately harm high protection areas is created. A novel management strategy would place the moderate protection reserves adjacent to areas that already have infrastructural development as a means of restricting further penetration into pristine areas. The moderate protection reserves with their close proximity to infrastructure would then help to insulate the more remote high protection reserves with a passive buffer in which no further infrastructural development could penetrate. Moreover, the establishment of such *insulation reserves would be more economically feasible since they would require less capital for road creation and maintenance as well as provide cheaper transportation to established economic markets.* Such an insulation protection system has not yet been published within or beyond the Amazon basin.

Indian lands represent 20% of the Brazilian Legal Amazon (Appendix C).

Although these reserves have received much attention and financial backing by the PP-G7 nations (\$340 million), they have yet to receive much attention from the conservation community. These lands prevent the greatest amount of degradation in heavily and moderately degraded habitats. These reserves react synergistically with lands that are heavily or moderately degraded by impeding their development beyond light degradation. Within the modeling scenarios, indigenous lands act passively to lands that are already classified as pristine, *i.e.* indigenous land designation does not change the habitat quality of the forest. Not surprisingly, they did not have influence on the spatial distribution of *pristine core forests.* *Indigenous areas are the single largest reserve management system that influences the distribution of fragmented forests.* These reserves alone are

responsible for 50% of the non-fragmented forests preserved by all Brazilian Legal Amazon reserve systems. The federal entity managing these reserves does not wish to further develop any relationships with either non-contacted tribes or tribes that have had adverse effects because of the influence of western contact (Tierney 2001). This reserve management strategy would seem to work in the cases of several western and northern Amazonian tribes that aggressively defend their territories from outsiders (Wallace 2003). This management technique fails for those people that have already suffered from the introduction of foreign diseases, food shortages and resource depletion from *surrounding western development* (Wallace 2003). *More aggressive interaction is* necessary not only to secure the land but also to preserve the actual people. Similar to the biological species diversity, the Amazon basin also holds a great diversity of indigenous human cultures that have evolved in analogous fashion (Meggars 1975). The Amazon basin contains the largest number of threatened language systems of any region of the world with an estimated 3 languages becoming extinct per year (Hinchberger 2002). The interest in saving biodiversity within the Amazon is also an act of saving human races from extinction through assimilation.

CONCLUSIONS

Road improvement within the Brazilian Legal Amazon has the greatest effects on deforestation of all of the types of projects within *Avaça Brasil*. Other infrastructural development projects are minor compared to those within *Avança Brasil*. One of the easiest ways to reduce habitat degradation, deforestation, and fragmentation is to spatially concentrate and overlap infrastructural projects such that penetration into inaccessible forests is minimized but local impacts are amplified within the development zone. Moderate protection reserves insulating these overlapped development areas would greatly reduce further incursion into adjacent forests limiting future deforestation events. Carbon trading mechanisms in the Kyoto Protocol have the potential to provide financial incentive to modify infrastructural development projects. This would reduce green house gas emissions incurred *via* forest destruction. With current development plans in place, the Brazilian Legal Amazon is likely to lose an additional 105,777 km² of pristine forest habitat by 2020. The loss of these forested systems is likely to increase carbon emissions from deforestation by a minimum of 20%.

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CONCLUSIONS

These studies have demonstrated how consideration of spatio-temporal interactions between forest dynamics and anthropogenic disturbance is critical for evaluation of forest integrity and appropriate conservation. Studies within this dissertation investigated the successional processes of hyper-diverse forest dynamics and the effects of anthropogenic tenure systems on carbon budget of total above-ground biomass (TAGB). Further, future deforestation, fragmentation, and habitat degradation effects were estimated under conditions of imminent infrastructural development within the Brazilian Legal Amazon currently proposed by the Brazilian government.

The combined, multi-media use of data capture technologies and intensive field sampling in this research yielded insightful results that would otherwise not have been possible utilizing ground information alone. While the partnered use of data-capture and field data has now become commonplace, an important aspect of the present research was testing and evaluating the sagacity of increased reliance on remote methods of data collection. The results here demonstrate that data-capture technologies provide a necessary and dynamic dimension to the analysis of large-scale ecological processes because of their now intensive and extensive coverage capabilities.

The central objective of the studies was to develop an understanding of the spatio-temporal aspects of ecological processes affecting tropical secondary forest regrowth and the dynamics upon which forest ecosystems depend for recovery. This entailed the collation of multi-scale data and analysis of data pertaining to mature forest structure, species richness, and integrated species relationships. By accounting for forest properties through time using data-capture models built on gathered spectral information, a

chronicle of the interplay of human land-use practices, mature forest spatial distributions, and secondary forests was developed. The resultant chronicle more accurately maps the estimated secondary forest contributions to greenhouse gas emissions within a complex and dynamic landscape than previous descriptions. Carbon tracking processes by this method are revealed to be more complex than previously assumed. Land tenure dynamics and heterogeneity preclude the use of simple static models. Study results imply that modeling the world's largest rain forest's effects on global greenhouse gas concentrations is highly uncertain and must be evaluated cautiously. By mapping and establishing the nature of the underlying ecological processes upon which secondary forests rely, and documenting how these forests' spatial distributions (in the vertical and horizontal) have changed over twenty years, the ability to predict future land development scenarios and their effects is much improved.

Although the theme running through all three manuscripts is the study of spatial distributions and patterns, the equally important theme of temporal variation is also present. Data-capture methods allowed for the tracking of temporal change of forest pattern from ground to canopy over large spatial extents. Time was accounted for by estimates of mature forest deforestation, secondary forest development, and subsequent conversion to agriculture occurring at annual intervals. Tracking the history of a pixel cell allows for an accurate estimation of secondary forests biomass, emissions, and decomposition. The overall results of these studies underscore the necessity of long-term and large-scale planning of Amazon land-use that takes such complexity into consideration. With comparable models, analyses, and methodologies developed here,

much more reliable estimates can be made that offer a clearer representation of future scenarios.

FINDINGS OF LANDSCAPE CHARACTERISTICS INFLUENCING SECONDARY FOREST TREE SPECIES RICHNESS

This study shows the importance of maintaining forests that contain the full biologic integrity for the persistence of these highly diverse forest systems. These ecological necessities dictate the sustainability of forest practices throughout the forested Amazon. In particular, this research identified and compared naturally occurring and anthropogenic disturbances and their effects on landscape function. For example, 'blowdown' events as large as 70 ha. within the central Amazon were identified. The size of these larger disturbances are roughly the same size as the majority of 'swidden' agricultural tenure systems but not of the scale of large favendas. However, their function is quite different due to the difference in the ways and location in which they are produced. Secondary forest's tree species richness is dependent on two factors, the architecture of the forest (age) and the proximity to seed source (large forests with frugivores). Once 10 years has past a higher number of tree species can establish in an environment of dampened micro-climatic variability. A viable seed source is not just an adjacent forest being able to provide seed but is also dependent on that seed's proper dispersal and germination. Step-wise multiple linear regression analysis has shown that the distance to continuous forest is the most significant factor in determining tree species richness indicating the importance of having viable seed dispersal vectors

A truly sustainable forestry practice and management strategy would need to use such findings in their management practices. Any management plan that does not consider the needs of replenishment will not retain the ability to sustain itself in perpetuity. Unfortunately preserving large continuous tracts of undisturbed forest, e.g. >

10,000 ha., often exceeds the size of the actual forest reserve. As a reference, Reservo Ducke is approximately 10,000 ha in size and is likely to lose its largest frugivore, tapirs (*Tapirus terrestris*), due to poaching and reduced population sizes. It should be noted that Reservo Ducke is an established and well published tropical forest research center, the premier of tropical forest research institutions within the central Amazon. This indicated that even in the best of circumstances, the effects of large-scale forest fragmentation are severe.

Further investigations into the spatial relationships of secondary forests across the Amazon basin would advance understanding of how prevalent this behavior is across tropical rain forests. It might be assumed that as forests transition from an alpha rich diversity structure (all species are rare and widespread) to a beta diverse forest structure (where single species are confined to local edaphic conditions, are abundant but the soils are spatially limited) a breakdown in the dependence on animal based seed dispersal would be observed. In beta diverse systems, dispersal beyond the confines of the soil type would not result in greater fecundity since depositing the seeds in uninhabitable conditions would not result in the species establishment. Furthermore, the limited spatial distribution of the soil would likely not support the necessary requirements for the evolution of seed dispersal mutualisms (i.e. the plant investing metabolic resources into high caloric fruits where the animal vector is restricted from its cohorts allowing for allopatric speciation processes). This is not to say that beta-diverse systems are not capable of mutualist relationships but that the mutualist relationships that they do develop will be of a smaller scale (e.g. insect as opposed to mammal) and not necessarily centered on seed dispersal since the species is confined to local soil conditions.

FINDINGS OF DEFORESTATION, LAND TENURE, FIRE AND INTER-ANNUAL VARIATION'S EFFECTS ON THE CARBON BUDGET

The methodology created for this research has made a good first-step in addressing the need for spatially explicit models of land tenure within tropical forested systems (Houghton et al. 1999, DeFries et al. 2002). This model is significantly more capable of accurately accounting and estimating TAGB occurring within a dynamic landscape than what could have been achieved by multiple ground measurements. The advance is largely because of the model's ability to census large areas on an annual basis. The model produces results that were significantly to highly significantly different when compared to extant models based on forest/non-forest distinctions. This implies that the spatial distribution of secondary forests and land tenure can have dramatic effects on the carbon budgets as well as tropical mature forest's contribution to global greenhouse gas emissions. Overall, the intensity of land tenure occurring in the southern tracts was estimated to have caused emissions that overwhelmed all sequestration occurring in areas of more conservative practices occurring in the north. Previous models of inter-annual variability have used the proportions of lands present in agriculture versus the proportion abandoned. This study illustrates how these assumptions are not realistic at portraying the actual dynamic contribution of these forests. The annual imagery census is capable of detecting differences in carbon sequestering landscapes that have more time between agriculture abandonment and re-establishment. This mapping indicates that the data-capture methods can accurately tract biomass changes occurring on the ground through time.

In terms of implications for future research, the methodology developed promises a good grounding for further investigation into the effects of landuse development that is rapidly changing the Brazilian Amazon. For example, this study is now being used to investigate the effects of road improvement along highway BR-174 that bisects the study area. The road was paved in 1997 and rapid development into previously inaccessible areas has already occurred. By extending the study beyond 1999, we will be more accurately able to predict the effects of road improvement on mature forest distribution, habitat degradation, and habitat fragmentation. Another important area of future study would be to replicate the analysis in key locations throughout the Brazilian Legal Amazon. With the replications in key locations, researchers would be able to more accurately extrapolate the findings to pan-Amazon. With such a study, 90% accuracy in TAGB dynamics across the Brazilian Amazon would be realized. This methodology also is very economically efficient to maintain through time. Continuing to account and census study areas in perpetuity is feasible and has the added feature that it can locate within 50m where deforestation is occurring on a yearly basis. At present, the resolution at the commensurate spatial extent is not possible with INPE's current methods. By being able to accurately locate where deforestation is occurring on an annual basis, great improvements in the effectiveness of government forest policies and management could reduce the rate of illegal clear-cut harvesting.

**FINDINGS OF CAUSES AND PREVENTION OF HABITAT DEGRADATION,
FOREST FRAGMENTATION AND DEFORESTATION IN THE BRAZILIAN
LEGAL AMAZON**

At present, large scale development projects within the Brazilian Legal Amazon have predicted that only 45 to 5% of the forested region will remain in pristine or pre-Columbian condition (Laurance et al. 2001a). By deconstructing the OPTIMISTIC build out scenario, it was possible to estimate how much area would be attributed to different types of development and evaluate how much forest would be protected with current reserve systems in place. The most destructive Avana Brasil project types were road upgrades, utility corridors, hydroelectric dams, new railroads and river channelization, in order of most to least. Road upgrades were also the major source for the loss of pristine forest habitats. Beyond the confines of the Avana Brasil project, riverside deforestation poses the largest potential threat due to the extent of navigable rivers within the watershed. With reserves in their present placement, 151,777 km² of heavily degraded and 277,269 km² moderately degraded forest habitat would be prevented. Much of the deleterious effect of the Avana Brasil could be prevented if projects in proximity to one another, are grouped together within the same localized area. For example, if road upgrades, new railroads and utility corridors were to be localized into one rather than three separate projects penetrating previously undisturbed forest. Presumably coalescing projects would also offer substantial economic savings due to decreases in the cost of deforesting areas.

The Element Added Model (EAM) and the Element Deleted Model (EDM) modeling scenarios proved to be an effective way to parameterize the potential direct and indirect effects of development project types in a complex landuse situation where many

projects can overlap within a small area. This type of vector based methodology works well for large-scale predictions (e.g., $> 1,000,000 \text{ km}^2$) but it is uncertain whether it can as well at smaller spatial scales. For regions of special importance it should be used in order to grasp a more holistic evaluation of the forest health of an area. Multiple smaller-scale impact statements on particular development projects provide for the evaluation of each singular project but ignore synergistic effects from combined development sources. In the Amazon's hyper-diverse flora and fauna with its complex interspecies relationships and dependencies, death from several small, almost benign, environmental degradations can easily degrade the forest to the extent where it is no longer capable of replacing itself.

POLITICS OF AMAZON CONSERVATION

Most of the public outcry over Amazonian deforestation comes from international concerns. Brazilians realize that the Amazon, as legally delineated, represents 58% of the nation's land holdings. Most of the major population centers within Brazil are situated along the eastern coast of the South Atlantic Ocean. It has only been since the 1960's that the Brazilian Government has actively pursued development within the interior Amazon Basin. Significant deforestation started with the construction of the Belem to Brasilia highway in the early 1960's. Previous to government sponsored infrastructure projects such as the construction of major roadways into the interior, the rainforest portion of the Amazon supported its indigenous peoples at population densities that were lower than the Saharan desert (Meggars, 1975). There are many reasons for this low population density, ranging from a low equilibrium balance due to the prevalence of parasites (e.g. if population densities get too high then parasites will find this 'source' and exploit it) to the low density of adequate edible protein set for human consumption (e.g. higher populations were viable near river courses but strictly terrestrial populations hold low population densities). In a sense, the costs of living within the Amazon Basin were great enough to delay the incursions of 'manifest destiny' that had occurred in North America and to a lesser degree in Eurasia.

Today, however, the previously inaccessible Amazon interiors are targets for their resources at a boundless rate and indigenous populations are being replaced with international and European-dominant populations. Presently, 12% of the Amazon Basin exists as habitat that is non-forested. Approximately 15% of this area has been deforested. The rates of deforestation had accelerated in the early millennium and have

stabilized at a rate that was 40% higher than what had occurred in the 1990's (INPE 2004). Population growth that is occurring within the Amazon Basin is an urbanized growth with populated economic centers expanding and peripheral villages senescing (Browder and Godfrey 1997). In large part, the urbanization derives from populations from peripheral towns that have collapsed once the local resources were exhausted (e.g. once that the valuable and marketable mahogany trees are harvested or gold cannot be recovered economically). Other peoples have been translocated to the Amazon being from the landless peasants class or small scale farmers of the south who have had their farms traded to larger agri-businesses. Some are from indigenous tribes that have been devastated by the introduction of new diseases and with the help of missionaries are trying to stay alive in a completely foreign society. In the southeastern Amazonia, there is building, persistent pressure from agricultural interests. Beef cattle once not-profitable without government subsidy are now profitable due to decreases in transportation costs (road improvements), globalization, and the exportation of beef to Asia (since the recent outbreak of bovine spongiforme encephalitis in the Pacific Northwest, U.S. beef exports are not allowed). With the recent devaluation of the Brazilian Real (R\$), beef exports to Asia and Europe are on the increase. Beyond beef, soybean cultivars genetically engineered to thrive in wet tropical environments are also a source of increased pressure to deforest. Manaus, once the center of rubber production in the entire world, is now a free trade zone with an economic base increasingly dependent on the export of manufactured goods. At present Sanyo, Phillips Electric, Vespa, Harley Davidson and other multi-national companies export their manufactured goods from Manaus to global markets.

Conservation strategies applied to the Amazon require just as much diversity as its ecosystems and the spectrum of economic pressures driving deforestation and habitat degradation. Today, 29.4% of the Brazilian Legal Amazon is under some form of conservation easement. High Protection reserves, where deforestation and habitat degradation is kept to a minimum, comprise 4.4% of the Brazilian Legal Amazon. Moderate protection reserves that allow for the limited harvest of flora and fauna comprise only 8.1%. There is one other major source of protection, however, that is quite substantive. Indigenous lands administered by the Brazilian Federal Government do not allow the take of any natural resources for sale comprise almost 20.3% of the Brazilian Legal Amazon. By comparison to northern countries such as the United States, this is a large proportion of national territory to be held in reserves; only the state of Alaska and Canada's Northwest Territory match such proportions. No other nation matches the actual absolute amount of land held within reserves at present (i.e., 2004). While this and other ecological research on the Amazon clearly support the termination of further development and urgent amelioration of extant environmental degradation, it is unlikely that the government of Brazil will respond in kind.

First, while much of the research is collaborative between Brazil and Euro-American countries like the U.S.A., there is a certain reserve and, in many cases, resentment by Brazilian nationals particularly in the government when such advice is conferred by the U.S. and members of the EU. These European states are not held as an exemplar in the area of conservation. The once biodiverse and species rich ecosystems of North America and Europe are themselves depauperate. For example, no comparable measures were taken in the U.S. to save 30% of the continental United States with 20%

being held within indigenous reserves as exists today in Brazil. If such measures had been taken early on in the colonial development, there would most likely be retained a far larger representation of North America's ecosystems with full function, e.g. great plains with bison, north Atlantic with large whales, eastern deciduous forest with old trees and large carnivores. In Brazil, there are severe internal population pressures, human poverty, and political-economic demands that easily distract from arguments that stress the need for conservation based on the merits of biodiversity.

What does remain salient amid these heated debates is that the Amazon is indeed unique in the world for its disproportionately high biodiversity. However, this unique diversity garners most of its appeal from its economic promise. The Amazon's biodiversity is a form of natural capital particularly in the pharmaceutical and cosmetic industries. Knowledge of the forest and its pharmaceutical potential is held by peoples that are remote and far from western economic influences. Until recently, most indigenous tribes have lived apart from Euro-American cultures. The rapid incursion of development, however, has sadly decimated non-human as well as human diversity. Indigenous peoples in their susceptibility to the domination of Euro-American culture, economics (e.g., money-based economies), and disease, are disappearing at an ever-increasing rate. Thus, some of the economic arguments that support conservation of the Amazon biodiversity at the same time undermine the viability of indigenous cultures and people. Often the only source to capitalize on this vast biocomplexity within the Amazon is the indigenous peoples that live within the forests. Ethnopharmacology has become yet another form of colonial exploitation and destruction. Now, the Brazilian

Government is in the final stages of construction of a research laboratory looking directly at bio-prospecting Amazonian species.

If the hyper-diverse forests of the Brazilian Amazon are to be conserved indefinitely, a re-evaluation of present reserve designs needs to be conducted and based on the findings of the studies of forests. Present reserve designs are based upon the findings of the Pleistocene refugia hypothesis, which after 30 years of investigation has yet to find definitive or conclusive evidence that the central Amazon was a savanna (Haeffer 1969, Colvinaux 1998). If reserves were to be designed based on the distribution of tree species (i.e. tree species centers of richness) the underlying habitat which faunal species depend upon and create mutualist relationships will most likely be conserved also. This is not to underscore the importance of fauna, but is based on flora that generally need more time to complete generation cycles. Reserves should be large enough to secure all tree species persistence indefinitely, meaning that the majority of species should have more than 500 pairs to avoid the possibility of inbreeding and genetic bottlenecks. After reserves establish persistent populations of tree species then the spatial configuration of the reserves should be amended to insure that they are large enough to meet the requirements of the larger seed dispersing fauna. Once the reserves are amended then considerations should be made to allow for bioprospecting, with the roads and rivers being 'patrolled' by these scientific endeavors. Indigenous populations should not be actively contacted due to their epidemiological sensitivity to foreign pathogens.

The conservation of hyper-diverse forests within the Amazon Basin is not merely a simple exercise of drawing boundaries around centers of endemism. It has to involve the local economics of diverse populations across the basin. Indeed, conservation

practice needs to reflect current ecological models that envision humans and nature as one interacting and interlocked system (Gunderson and Holling, 2001). Without such simultaneous consideration of indigenous and ecological conservation, human and nonhuman nature will be comprised and lost. Indigenous populations are especially susceptible to diseases introduced by good intentioned scientists. These peoples should be left alone to live as they have for the past 5,000 years , as is the policy of the Brazilian Government towards tribes that have not yet been contacted. Self-rule needs to be a principle that is held for all peoples and is defined as part of conservation itself. Management within reserve systems also needs to be based on scientific inquiries of biocomplex systems not just single species studies: studies based on suitable harvest regimes or maladapted management from the temperate north. The gift of comparative studies demonstrates that attention to specificity is essential for guiding conservation strategies and design. Subsequently, more localized and indigenous science is needed in order to secure the longevity of these forests that hold a third of all the species on earth (Wilson 1988).

Finally, there is one critical "take home message" for scientists abroad conducting research in the Amazon and elsewhere including the United States. Human population increases and consumption need to be regarded as a problem of biophysical science as well as social and public debate. These two factors remain the fundamental source for creating the decimation of the Amazon and impeding successful conservation. Science cannot be effective without including this explicitly in the way in which research is designed and inference is made.

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Appendices

APPENDIX A STATISTICAL ANALYSES FOR CHAPTER TWO

Table A.1 Index of tree species and families encountered in the 29 100 m transects.

1	<i>Annona ambotay</i> Aubl.	Annonaceae
2	<i>Guatteria foliosa</i> Benth.	Annonaceae
3	<i>Bocageopsis</i> sp.	Annonaceae
4	<i>Guatteria discolor</i> R.E. Fries	Annonaceae
5	<i>Onychopetalum amazonicum</i> R.E. Fries	Annonaceae
6	<i>Rollinia eszucca</i> (DC. ex Dunal) A. DC.	Annonaceae
7	<i>Xylopia benthamii</i> R.E. Fries	Annonaceae
8	<i>Anacardium microsepalum</i>	Anacardiaceae
9	<i>Tapirira guianensis</i> Aubl.	Anacardiaceae
10	<i>Couma utilis</i>	Apocynaceae
11	<i>Geissospermum argenteum</i> Wood	Apocynaceae
12	<i>Lacmelia gracilis</i> Wood	Apocynaceae
13	<i>Astrocaryum mumbaca</i> Mart.	Arecaceae
14	<i>Euterpe precatoria</i> Mart.	Arecaceae
15	<i>Oenocarpus bacaba</i> Mart.	Arecaceae
16	<i>Syagrus inajai</i>	Arecaceae
17	<i>Jacaranda copaia</i> (Aubl.) D.Don.	Bignoniaceae
18	<i>Tabebuia serratifolia</i>	Bignoniaceae
19	<i>Ceiba</i> sp.	Bombacaceae
20	<i>Cordia geoldiana</i> Huber	Boraginaceae
21	<i>Protium</i> sp.	Burseraceae
22	<i>Hymenaea courbaril</i> L.	Caesalpiniaceae
23	<i>Peltogyne</i> sp.	Caesalpiniaceae
24	<i>Sclerolobium</i> sp.	Caesalpiniaceae
25	<i>Swartzia</i> sp.	Caesalpiniaceae
26	<i>Caryocar villosum</i> (Aubl.) Pers.	Caryocaraceae
27	<i>Goupia glabra</i> Aubl.	Celastraceae
28	<i>Cecropia sciadophylla</i> Mart.	Cecropiaceae
29	<i>Cecropia leucocoma</i> Miq.	Cecropiaceae
30	<i>Cecropia</i> sp.	Cecropiaceae
31	<i>Cecropia purpurescens</i> C.C. Berg.	Cecropiaceae
32	<i>Cecropia ficifolia</i> Warb. ex Senth.	Cecropiaceae
33	<i>Pourouma mellinonii</i>	Cecropiaceae
34	<i>Pourouma tomentosa</i> C. Mart.	Cecropiaceae
35	<i>Couepia</i> sp.	Chrysobalanaceae
36	<i>Licania</i> sp.	Chrysobalanaceae
37	<i>Symphonia globulifera</i> L. f.	Clusiaceae
38	<i>Vismia cayennensis</i> (Jacq.) Pers.	Clusiaceae
39	<i>Vismia guianensis</i> (Aubl.) Choisy	Clusiaceae

Table A.1 Index of tree species and families encountered in the 29 100 m transects (Continued).

40	<i>Vismia japurensis</i> Reichardt	Clusiaceae
41	<i>Vismia duckei</i> A.C. Smith	Clusiaceae
42	<i>Tapura</i> sp.	Dichapetalaceae
43	<i>Sloanea</i> sp.	Elaeocarpaceae
44	<i>Apartisthmium cordatum</i> (A. Juss.) Baill.	Euphorbiaceae
45	<i>Croton lonjouensis</i> Jablonon	Euphorbiaceae
46	<i>Hevea</i> sp.	Euphorbiaceae
47	<i>Mabea</i> sp.	Euphorbiaceae
48	<i>Micrandropsis schleroxylon</i>	Euphorbiaceae
49	<i>Sapium</i> sp.	Euphorbiaceae
50	<i>Andira</i> sp.	Fabaceae
51	<i>Dipteryx odorata</i> (Aubl.) Willd.	Fabaceae
52	Unknown	Fabaceae
53	<i>Ormosia</i> sp.	Fabaceae
54	<i>Casearia</i> sp.	Flacourtiaceae
55	<i>Lindackeria</i> sp.	Flacourtiaceae
56	<i>Laetia procera</i> (Poep.) Eichl.	Flacourtiaceae
57	<i>Ryania speciosa</i>	Flacourtiaceae
58	<i>Sacoglottis</i> sp.	Houmiriaceae
59	<i>Aniba hostmaniana</i> (Nees) Mez	Lauraceae
60	<i>Aniba</i> sp.	Lauraceae
61	<i>Licaria</i> sp.	Lauraceae
62	<i>Ocotea</i> sp.	Lauraceae
63	<i>Eschweilera</i> sp.	Lecythidaceae
64	<i>Lecythis</i> sp.	Lecythidaceae
65	<i>Corythophora</i> sp.	Lecythidaceae
66	<i>Byrsonima crispa</i> Adr. Juss.	Malpighiaceae
67	<i>Bellucia grossularioides</i> (L.) Triana	Melastomataceae
68	<i>Miconia</i> sp.	Melastomataceae
69	<i>Miconia tomentosa</i> (Rich.) D. Don	Melastomataceae
70	<i>Miconia</i> sp.	Melastomataceae
71	<i>Miconia affinis</i> DC.	Melastomataceae
72	<i>Miconia argyrophylla</i> DC.	Melastomataceae
73	<i>Mouriri duckeana</i> Morley	Memecylaceae
74	<i>Guarea</i> sp.	Meliaceae
75	<i>Trichilia</i> sp.	Meliaceae
76	<i>Inga falcistipula</i> Ducke	Mimosaceae
77	<i>Inga heterophylla</i> Willd	Mimosaceae
78	<i>Inga velutina</i>	Mimosaceae
79	<i>Parkia</i> sp.	Mimosaceae
80	<i>Phithecellobium racemosum</i> Ducke	Mimosaceae
81	<i>Stryphnodendron pulcherrimum</i> (Will.) Hoch.	Mimosaceae
82	<i>Siparuna amazoniaca</i> Mart.	Siparunaceae
83	<i>Brosimum parinarioides</i> Ducke	Moraceae
84	<i>Ficus</i> sp.	Moraceae
85	<i>Naucleopsis</i> sp.	Moraceae

Table A.1 Index of tree species and families encountered in the 29 100 m transects (Continued).

86	<i>Pseudolmedia</i> sp.	Moraceae
87	<i>Sorocea guillemmiana</i>	Moraceae
88	<i>Phenakospermum guyanense</i> (Richard) Endl.	Strelitziaceae
89	<i>Campsonura ulei</i> Warb	Myristiaceae
90	<i>Iranthera</i> sp.	Myristiaceae
91	<i>Virola</i> sp.	Myristiaceae
92	<i>Eugenia</i> sp.	Myrtaceae
93	<i>Myrcia</i> sp.	Myrtaceae
94	<i>Psidium</i> sp.	Myrtaceae
95	<i>Neea</i> sp.	Nyctaginaceae
96	<i>Heisteria</i> sp.	Olacaceae
97	<i>Alibertia edulis</i> Rich.	Rubiaceae
98	<i>Palicourea</i> sp.	Rubiaceae
99	<i>Psychotria</i> sp.	Rubiaceae
100	<i>Zanthoxylum</i> sp.	Rutaceae
101	<i>Talisia cupularis</i> Radlk.	Sapindaceae
102	<i>Chrysophyllum oppositum</i>	Sapotaceae
103	<i>Chrysophyllum</i> sp.	Sapotaceae
104	<i>Micropholis mensalis</i> (Baeh.) Aubr.	Sapotaceae
105	<i>Pouteria</i> sp.	Sapotaceae
106	<i>Richardella manaosensis</i>	Sapotaceae
107	<i>Micropholis</i> sp.	Sapotaceae
108	<i>Simarouba amara</i> Aubl.	Simaroubaceae
109	<i>Apeiba echinata</i> Gaertn	Tiliaceae
110	<i>Leuhea</i> sp.	Tiliaceae
111	<i>Trema micrantha</i> (L.) Blume	Ulmaceae
112	<i>Aegiphila integrifolia</i> (Jac.) Jac.	Verbenaceae
113	<i>Rinorea</i> sp.	Violaceae
114	<i>Rinorea guianensis</i> Aubl.	Violaceae
115	<i>Erisma</i> sp.	Vochysiaceae
116	Unknown	Unknown
117	<i>Cupania</i> sp.	Sapindaceae
118	<i>Theobroma</i> sp.	Sterculiaceae
119	<i>Scleronema micrantha</i>	Bombacaceae
120	Rubiaceae sp.	Rubiaceae
121	<i>Minquartia guianensis</i> Aubl.	Olacaceae
122	<i>Oenocarpus bataua</i> Mart.	Arecaceae
123	<i>Pithecelobium</i> sp.	Mimosaceae
124	<i>Euterpe</i> sp.	Arecaceae
125	<i>Inga speciosa</i> M. Mart. & Galeotti	Meliaceae
126	<i>Pourouma</i> sp.	Cecropiaceae
127	<i>Vismia cauliflora</i>	Clusiaceae
128	<i>Cybianthus</i> sp.	Myrsinaceae
129	<i>Erythroxylum</i> sp.	Erythroxylaceae
130	<i>Duroia</i> sp.	Rubiaceae
131	<i>Siparuna</i> sp.	Siparunaceae

Table A.1 Index of tree species and families encountered in the 29 100 m transects (Continued).

132	<i>Micrandra</i> sp.	<i>Euphorbiaceae</i>
133	Unknown	<i>Moraceae</i>
134	<i>Hirtella</i> sp.	<i>Chrysobalanaceae</i>
135	Unknown	<i>Apocynaceae</i>
136	<i>Posoqueria</i> sp.	<i>Rubiaceae</i>
137	<i>Phyllanthus</i> sp.	<i>Euphorbiaceae</i>
138	Unknown	<i>Myrtaceae</i>
140	<i>Lacistema</i> sp.	<i>Lacistemataceae</i>
142	<i>Byrsonima lancifolia</i>	<i>Malpighiaceae</i>
143	<i>Lecythis usitata</i>	<i>Lecythidaceae</i>
145	<i>Brosimum guianensis</i> (Aubl.) Ducke	<i>Moraceae</i>
146	<i>Byrsonima</i> sp.	<i>Malpighiaceae</i>
147	<i>Macrolobium</i> sp.	<i>Caesalpiniaceae</i>
148	<i>Talisia</i> sp.	<i>Sapindaceae</i>
150	<i>Rauwolfia</i> sp.	<i>Apocynaceae</i>
151	<i>Maprounea</i> sp.	<i>Euphorbiaceae</i>
152	<i>Dinizia excelsa</i> Ducke	<i>Mimosaceae</i>
153	<i>Bellucia imperialis</i> Saldanha & Cogn.	<i>Melastomataceae</i>
155	<i>Rinorea racemosa</i> (Mart.) Kuntze	<i>Violaceae</i>
156	<i>Rollinia insignis</i> R.E. Fries	<i>Annonaceae</i>
158	<i>Enterolobium schomburgkii</i> (Benth.) Benth.	<i>Mimosaceae</i>
159	<i>Ambelania tennifolia</i>	<i>Apocynaceae</i>
160	<i>Trattinnickia</i> sp.	<i>Burseraceae</i>
161	<i>Vitez</i> sp.	<i>Verbenaceae</i>
162	<i>Coussapoua</i> sp.	<i>Cecropiaceae</i>
163	<i>Pausandra</i> sp.	<i>Euphorbiaceae</i>
165	<i>Hymenolobium excelsum</i>	<i>Fabaceae</i>
166	<i>Caryocar pallidum</i>	<i>Caryocaraceae</i>
167	<i>Sterculia</i> sp.	<i>Sterculiaceae</i>
168	<i>Couratari stellata</i>	<i>Lecythidaceae</i>
169	<i>Ephedranthus amazonicus</i> R.E. Fries	<i>Annonaceae</i>
170	<i>Miconia myriantha</i> Benth. Ducke	<i>Melastomataceae</i>
171	<i>Anisophyllea</i> sp.	<i>Anisophyllaceae</i>
172	<i>Annona</i> sp.	<i>Annonaceae</i>
173	<i>Croton</i> sp.	<i>Euphorbiaceae</i>
174	<i>Trymatococcus</i> sp.	<i>Moraceae</i>
175	<i>Isertia</i> sp.	<i>Rubiaceae</i>
176	<i>Alchornea</i> sp.	<i>Euphorbiaceae</i>
177	<i>Lacunaria</i> sp.	<i>Quiinaceae</i>
178	<i>Swartzia corrugata</i> Benth.	<i>Caesalpiniaceae</i>
180	<i>Buchenavia</i> sp.	<i>Combretaceae</i>
181	<i>Simaba cedron</i>	<i>Simaroubaceae</i>
182	<i>Simaba guianensis</i> Cavalcante	<i>Simaroubaceae</i>
183	<i>Eperua bijuga</i>	<i>Caesalpiniaceae</i>
185	<i>Diospyros</i> sp.	<i>Ebenaceae</i>
186	<i>Clarisia racemosa</i> R. et P.	<i>Moraceae</i>

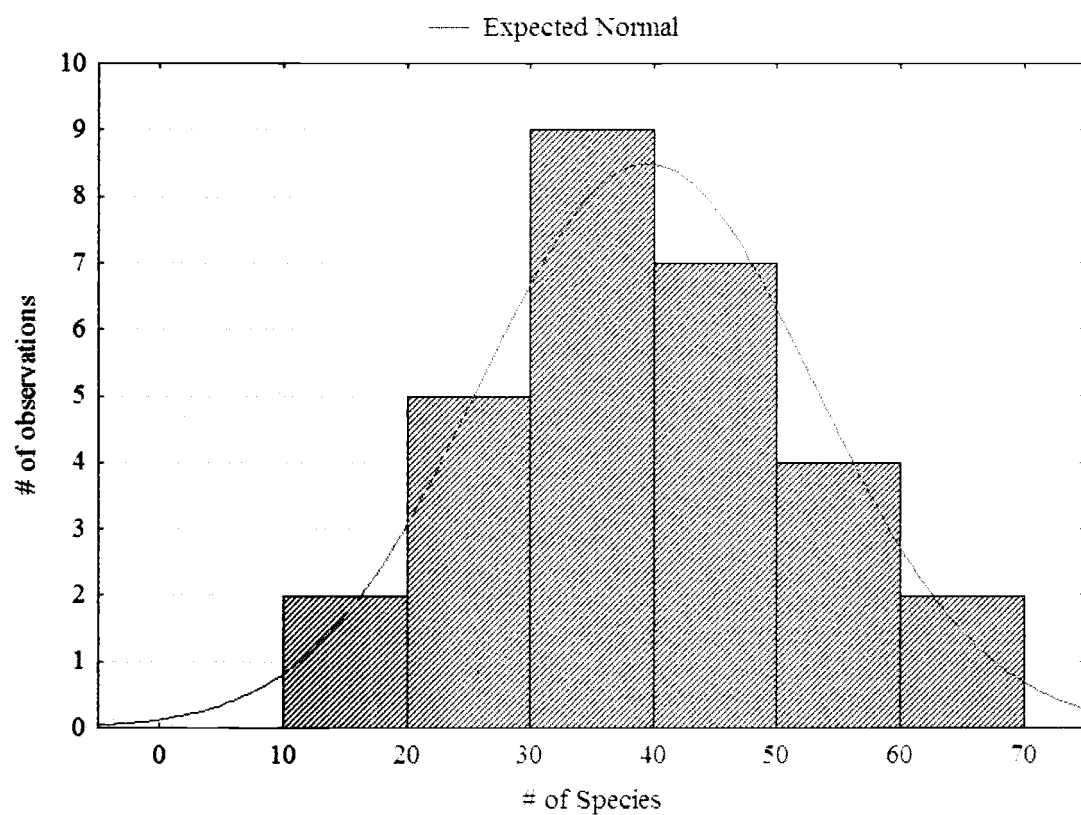


Figure A.1 Histogram of the Normality of the Dependent Variable Species Richness with an expected normal distribution overlay.

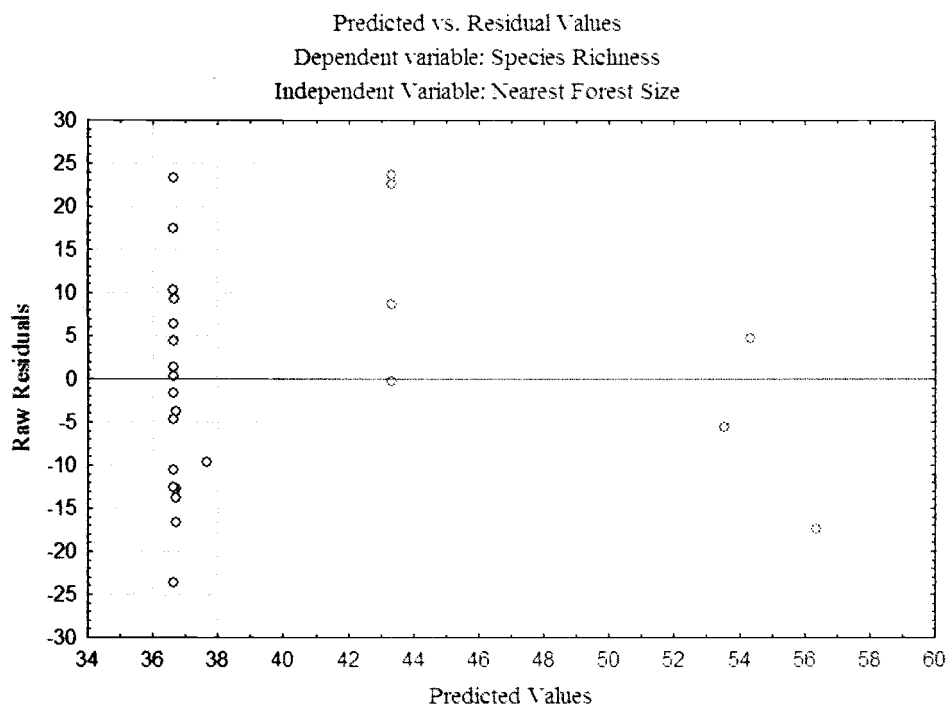


Figure A.2 Predicted *versus* residuals plot showing a horn shape indicative of non-normally distributed residual variances for the independent variable nearest forest size.

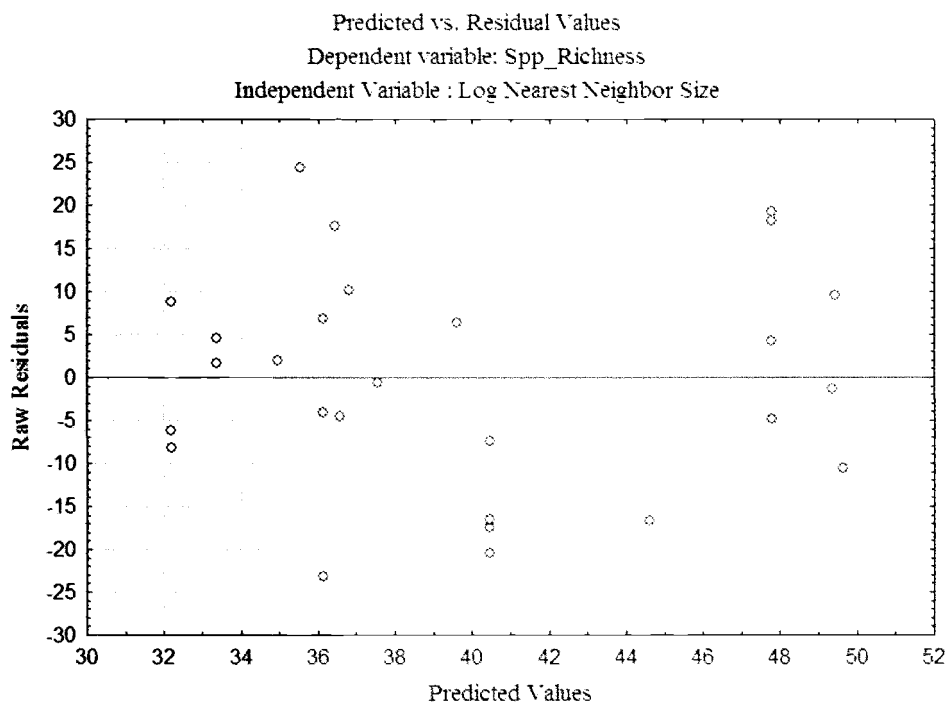


Figure A.3 Predicted *versus* residual plot showing normal distribution of the residuals for the independent variable Log nearest forest size.

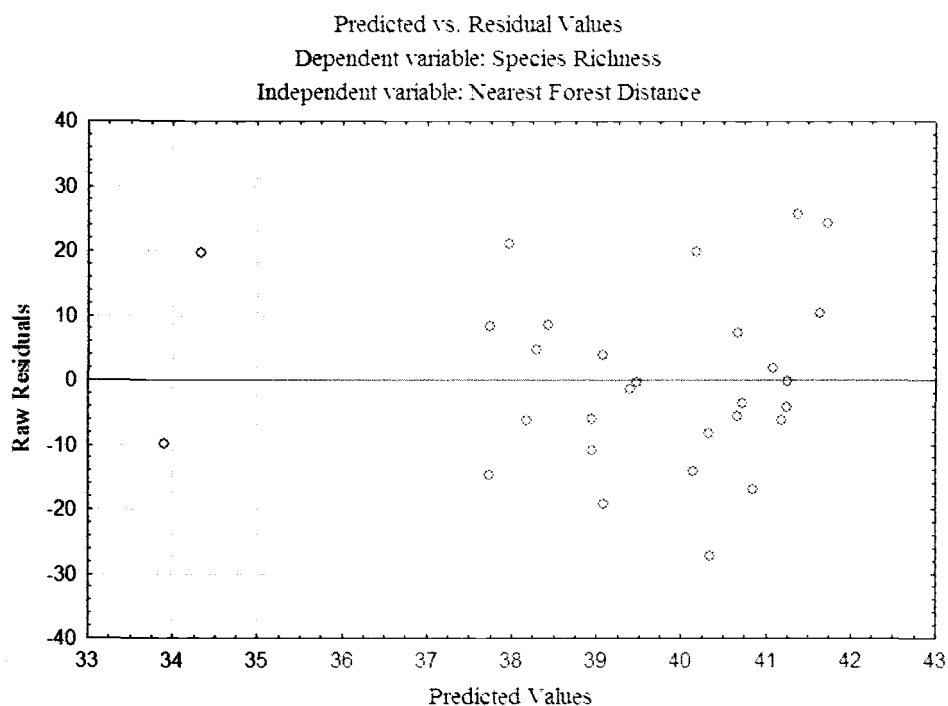


Figure A.4 Predicted *versus* residual plot showing a normal distribution of the residuals for the independent variable nearest forest distance.

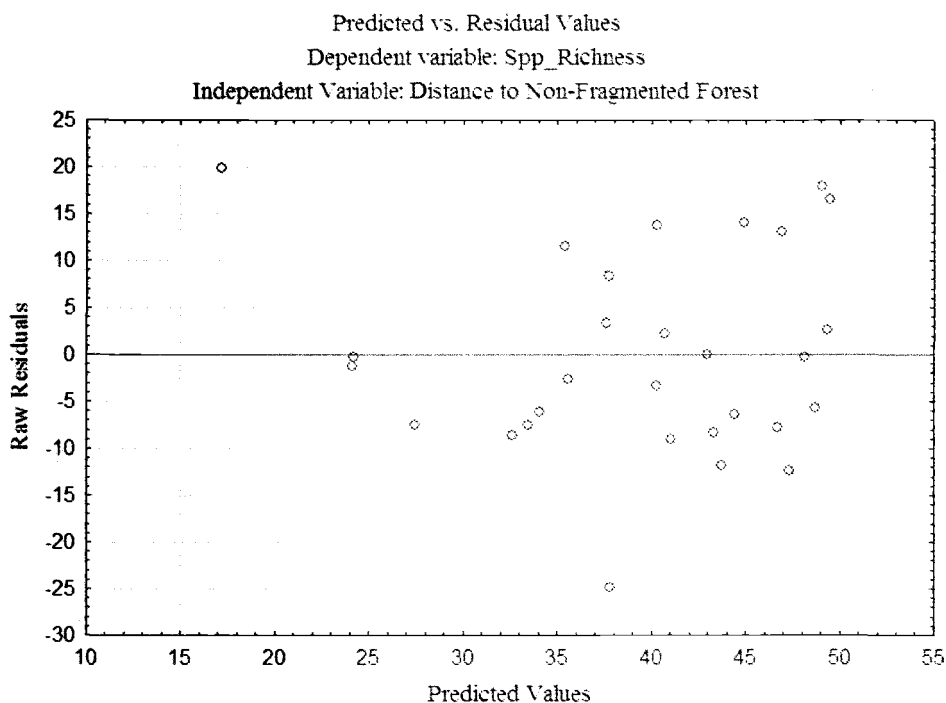


Figure A.5 Predicted *versus* residual plot showing a horn shape indicative of non-normally distributed residual variances for the independent variable distance to non-fragmented forest.

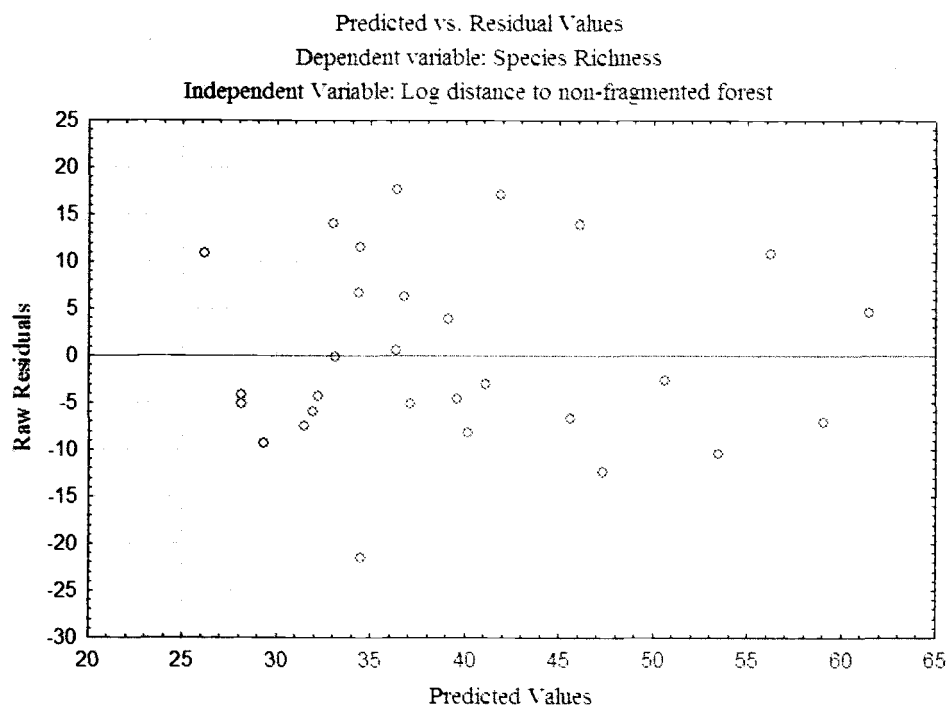


Figure A.6 Predicted *versus* residual plot showing normal distribution of the residuals for the independent variable Log distance to non-fragmented forest.

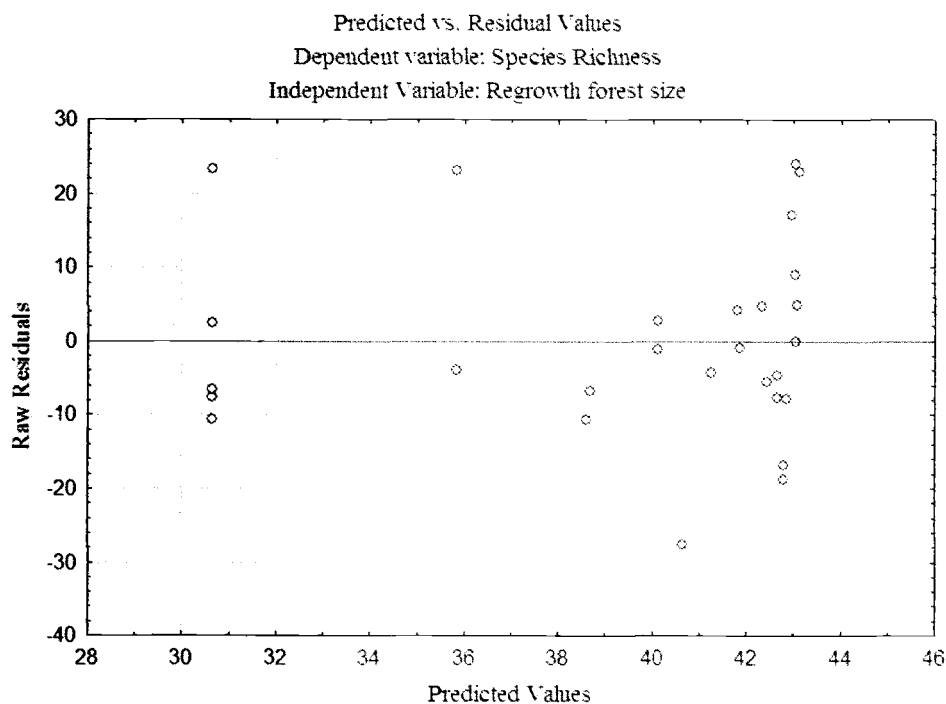


Figure A.7 Predicted *versus* residual plot showing a normal distribution of the residuals for the independent variable regrowth forest size.

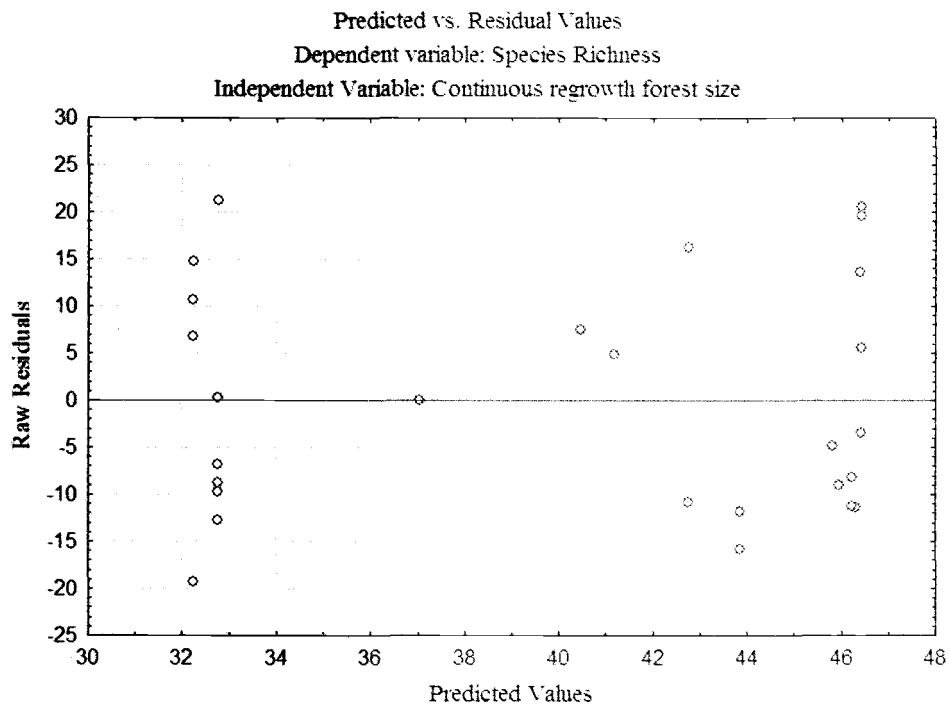


Figure A.8 Predicted *versus* residual plot showing a near normal distribution of the residuals for the independent variable continuous regrowth forest size.

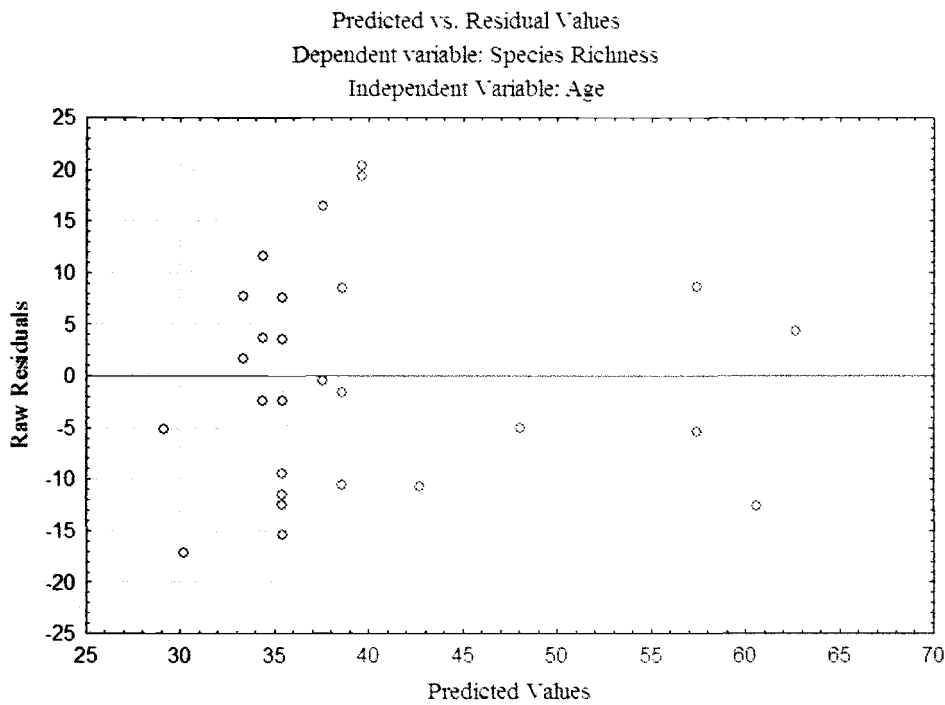


Figure A.9 Predicted *versus* residual plot showing a normal distribution of the residuals for the independent variable age.

Table A.2 Simple linear regression results of the dependent variable species richness *versus* the independent variable $\log_{(10)}$ nearest forest size.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	986.9197	986.9197	6.331938	0.018106	
Residual	27	4208.322	155.8638			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	32.19909	3.708513	8.682481	0.000000	24.58985	39.80833
Log nearest forest size	3.93992	1.565738	2.516334	0.018106	0.72729	7.15255

$$R^2_{adj} = 0.159965$$

$$n = 29$$

Table A.3 Simple linear regression results of the dependent variable species richness *versus* the independent variable nearest forest distance.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	104.8335	104.8335	0.556047	0.462301	
Residual	27	5090.408	188.5336			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	42.14091	4.382731	9.615217	0.000000	33.14829	51.13353
Nearest forest distance	-0.01632	0.021882	-0.745685	0.462301	-0.06121	0.02858

$$R^2_{adj} = 0.016111$$

$$n = 29$$

Table A.4 Simple linear regression results of the dependent variable species richness *versus* the independent variable $\log_{(10)}$ distance to non-fragmented forest.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	2525.816	2525.816	25.54747	0.000026	
Residual	27	2669.425	98.86759			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	88.58895	9.889335	8.95803	0.000000	68.2977	108.8802
Log distance cont. forest	-8.43002	1.667841	-5.05445	0.000026	-11.8521	-5.0079

$$R^2_{adj} = 0.467148$$

$$n = 29$$

Table A.5 Simple linear regression results of the dependent variable species richness *versus* the independent variable regrowth forest size.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	582.3988	582.3988	3.408911	0.075835	
Residual	27	4612.843	170.8460			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	43.18580	3.148618	13.71580	0.000000	36.72537	49.64623
Regrowth forest size	-0.01404	0.007602	-1.84632	0.075835	-0.02963	0.00156

$$R^2_{adj} = 0.079217$$

$$n = 29$$

Table A.6 Simple linear regression results of the dependent variable species richness *versus* the independent variable continuous regrowth forest size.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	1108.936	4086.305	7.327227	0.011631	
Residual	27	1108.936	151.3446			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	46.50230	3.455945	13.45574	0.000000	39.41129	53.59331
Cont. Regrowth forest size	-0.00717	0.002649	-2.70689	0.011631	-0.01260	-0.00173

$$R^2_{adj} = 0.184321$$

$$n = 29$$

Table A.7 Simple linear regression results of the dependent variable species richness *versus* the independent variable regrowth forest age.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	2241.271	2241.271	20.48576	0.000109	
Residual	27	2953.970	109.4063			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	25.99267	3.557526	7.306389	0.000000	18.69323	33.29211
Regrowth forest age	1.04883	0.231728	4.526120	0.000109	0.57336	1.52429

$$R^2_{adj} = 0.410350$$

$$n = 29$$

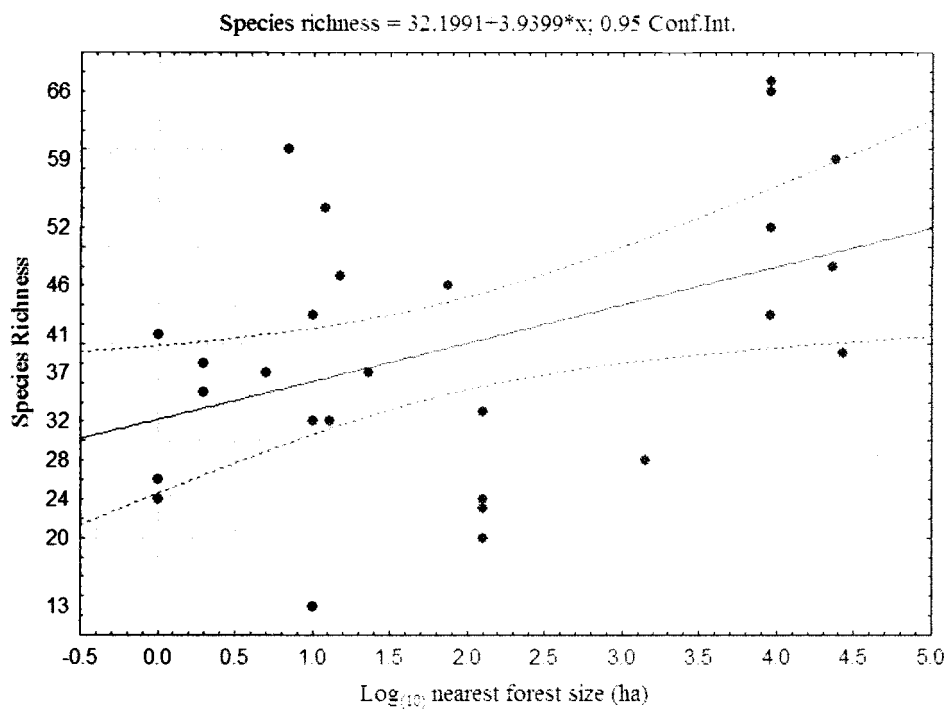


Figure A.10 Scatterplot for \log_{10} nearest forest size *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

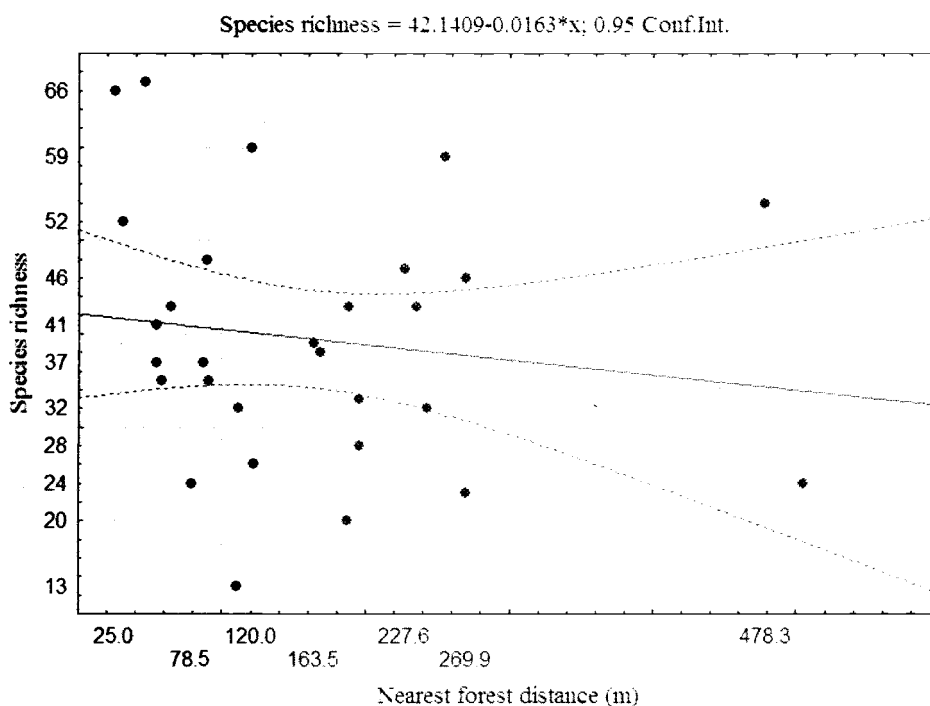


Figure A.11 Scatterplot for nearest forest distance *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

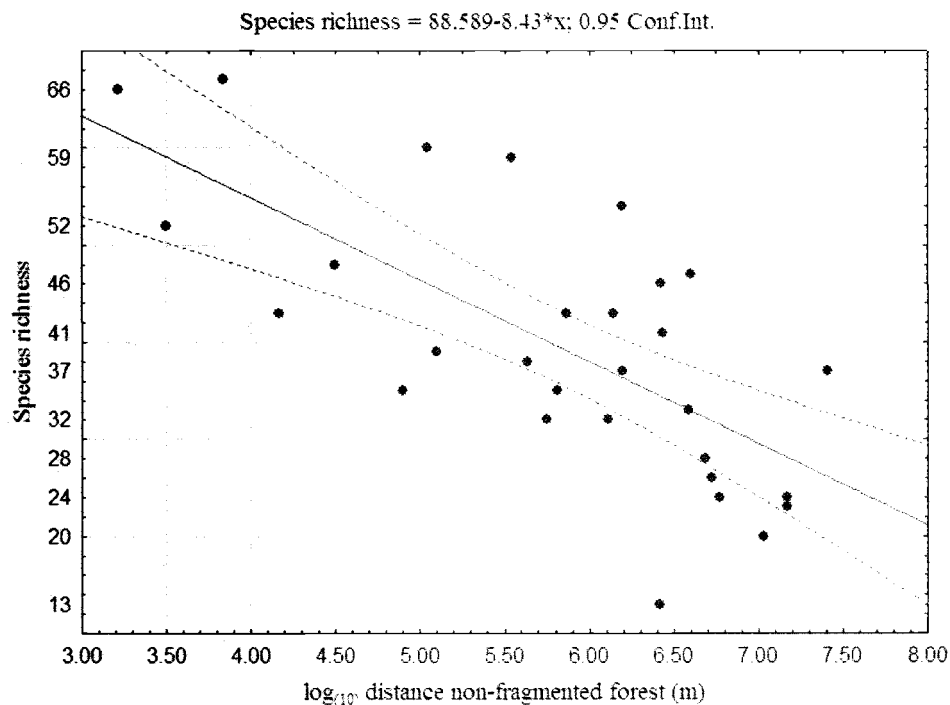


Figure A.12 Scatterplot for \log_{10} distance to non-fragmented forests *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

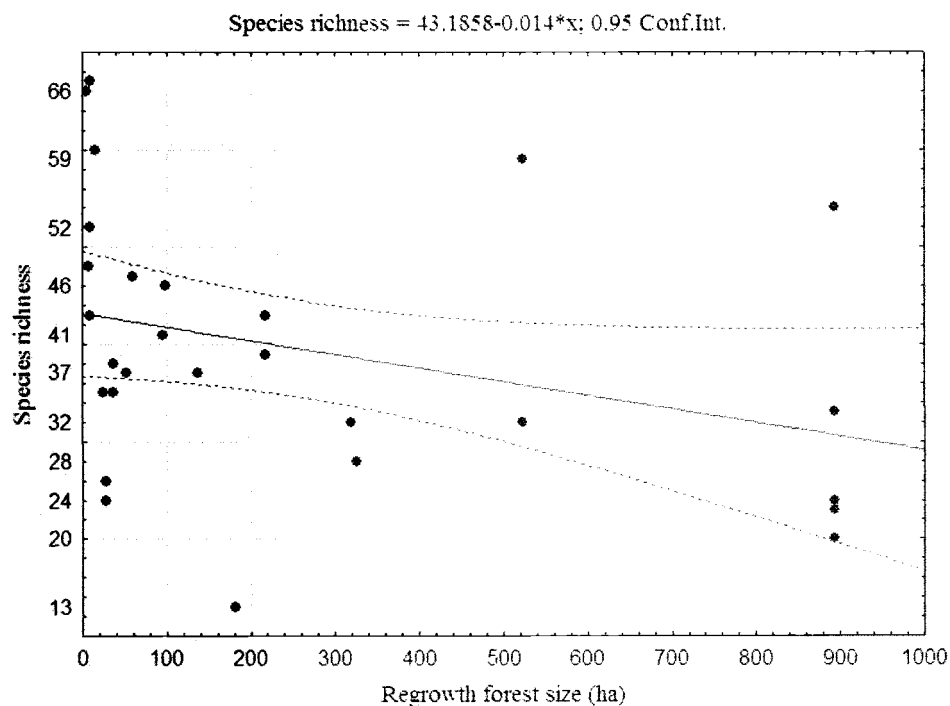


Figure A.13 Scatterplot for regrowth forest size *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

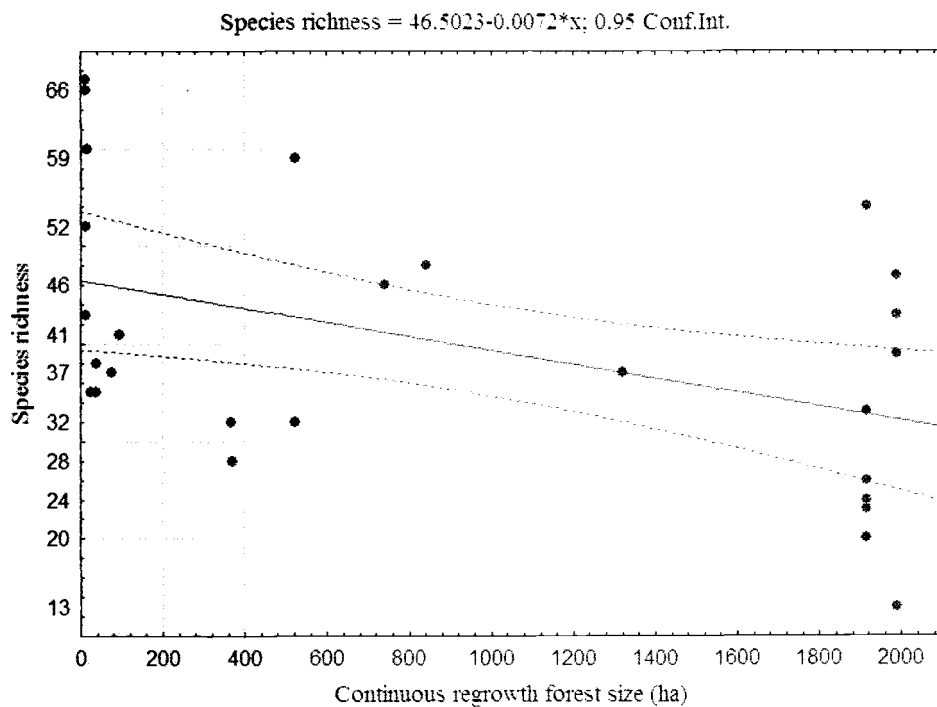


Figure A.14 Scatterplot for continuous regrowth forest size *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

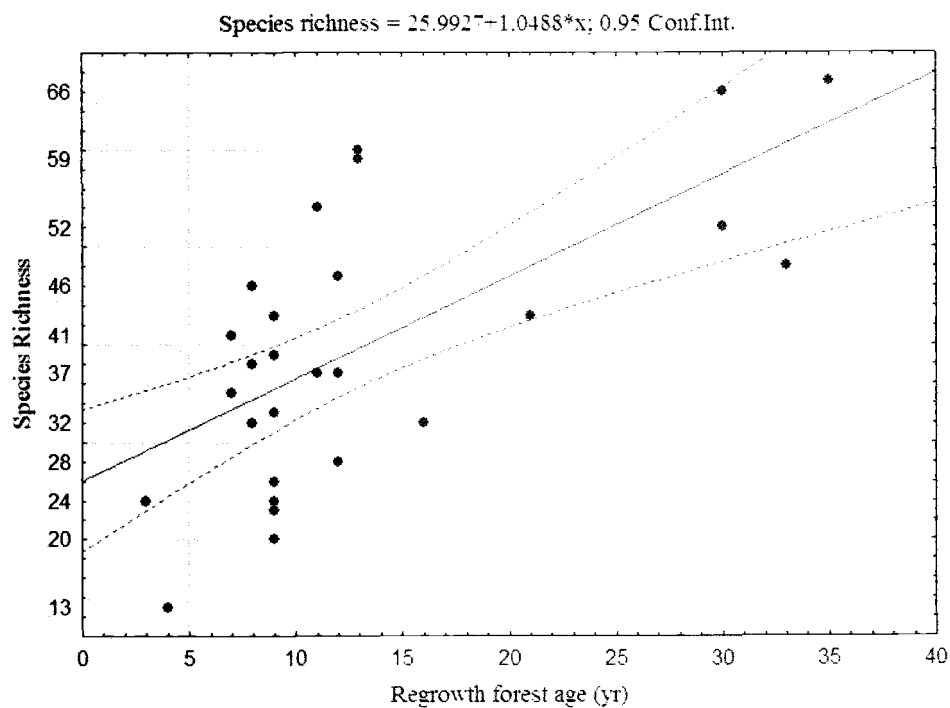


Figure A.15 Scatterplot for regrowth forest age *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line)

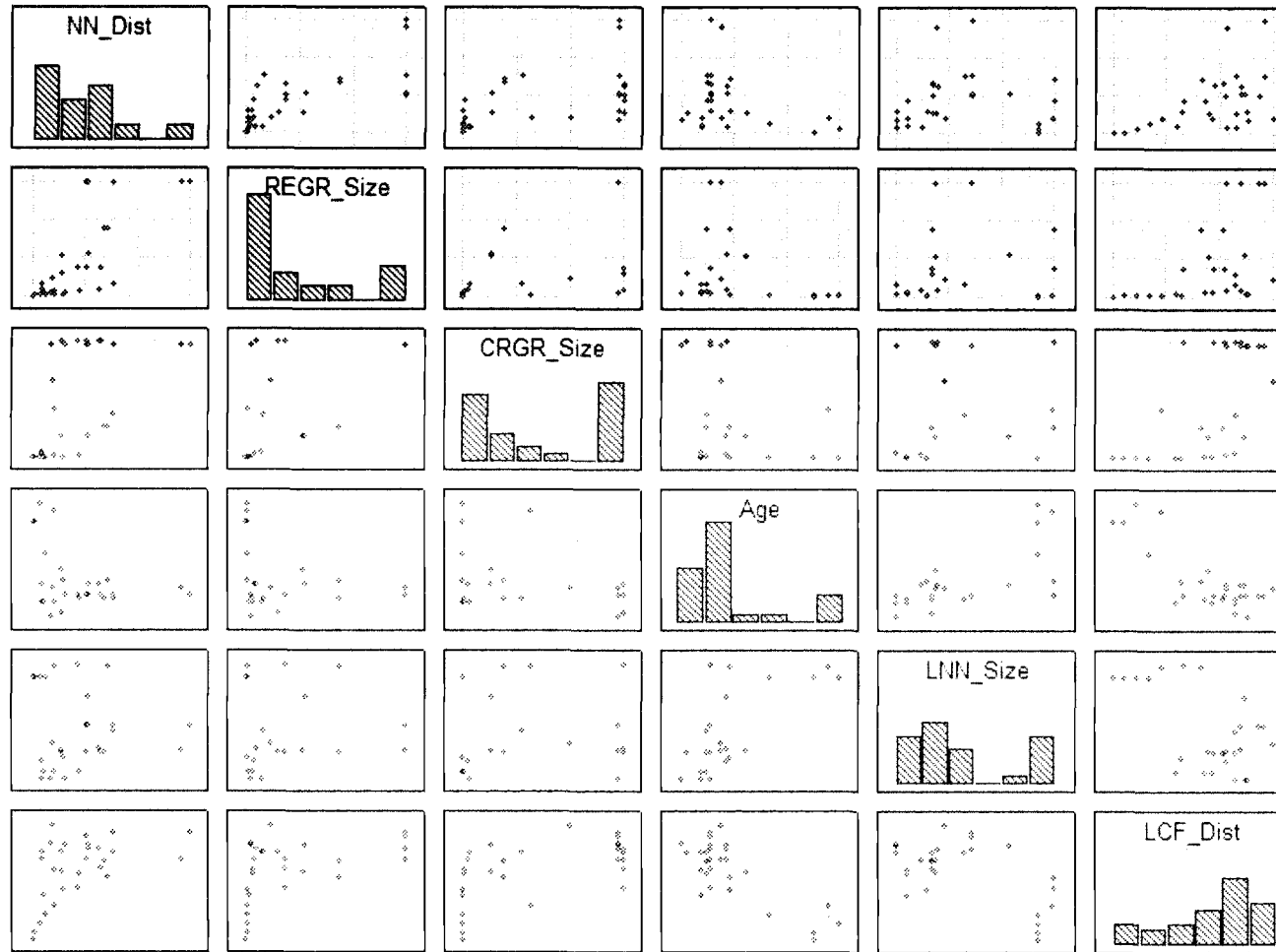


Figure A.16 Matrix scatterplot of the independent variables nearest forest distance, regrowth forest size, continuous regrowth forest size, age of regrowth forest, $\log_{(10)}$ nearest forest size and $\log_{(10)}$ distance to non-fragmented forest with histograms of these variables frequency distributions.

Table A.8 Summary statistics of the stepwise forward general regression model (GRM) for the effects of the independent variables, $\log_{(10)}$ nearest forest size, nearest forest distance, $\log_{(10)}$ distance to non-fragmented forest, regrowth forest size, continuous regrowth forest size and age of regrowth forest *versus* the dependent variable species richness.

Summary of stepwise regression; variable: Species richness
Forward stepwise
P to enter: 0.05, P to remove: 0.05

	Steps	Degr. of Freedom	F to remove	P to remove	F to enter	P to Enter	Effect status
Nearest forest distance	Step 1	1			0.55605	0.462301	Out
Regrowth forest size		1			3.40891	0.075835	Out
Cont. regrowth forest size		1			7.32723	0.011631	Out
Age of regrowth forest		1			20.48576	0.000109	Out
$\log_{(10)}$ Nearest forest size		1			6.33192	0.018106	Out
$\log_{(10)}$ Distance to non-frag. forest		1			25.54747	0.000026	Entered
$\log_{(10)}$ Distance to non-frag. forest	Step 2	1	25.54747	0.000026			In
Regrowth forest size		1			0.01081	0.917982	Out
Cont. regrowth forest size		1			0.11850	0.733434	Out
Age of regrowth forest		1			1.72419	0.200626	Out
$\log_{(10)}$ Nearest forest size		1			0.14577	0.705716	Out
Nearest forest distance		1			3.00624	0.094789	Out

Table A.9 Stepwise forward general regression model (GRM) results for the effects of the independent variables $\log_{(10)}$ distance to non-fragmented forest *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	2525.816	2525.816	25.54747	0.000026	
Residual	27	2669.425	98.86759			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	36.51866	5.343806	6.83383	0.000000	25.53431	47.50301
Distance to non-frag. forest	-19.4108	3.840346	-5.05445	0.000026	-27.2906	-11.5311

$$R^2_{adj} = 0.467148$$

$$n = 29$$

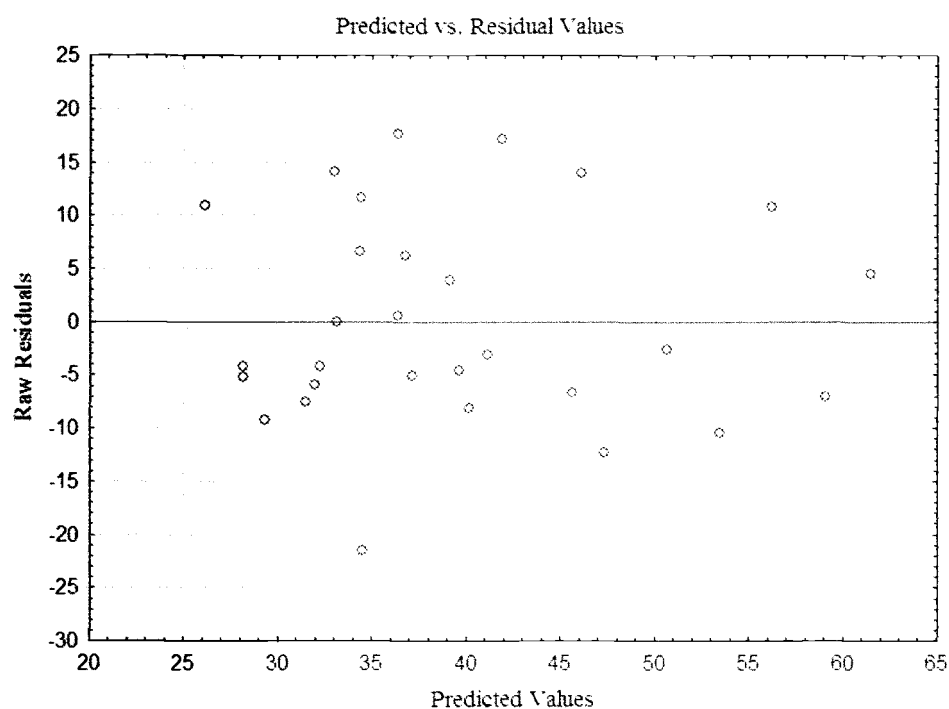


Figure A.17 Predicted *versus* Residuals plot showing a normally distributed residual variances for the independent variable $\log_{(10)}$ distance to non-fragmented forest *versus* the dependent variable species richness.

Table A.10 Summary statistics of the stepwise backward general regression model (GRM) for the effects of the independent variables, nearest forest size, nearest forest distance, distance to non-fragmented forest, regrowth forest size, continuous regrowth forest size and age of regrowth forest *versus* the dependent variable species richness.

Summary of stepwise regression; variable: Species richness
Backward stepwise
P to enter: 0.05, P to remove: 0.05

	Steps	Degr.Of Freedom	F to Enter	P to Enter	F to Exit	P to Exit	Effect status
Nearest forest Size	Step 1	1	0.08880	0.768506			Removed
Nearest forest distance		1	5.22839	0.032210			In
Distance to non-frag. forest		1	2.08574	0.162772			In
Regrowth forest size		1	2.18799	0.153268			In
Cont. regrowth forest size		1	0.83720	0.370120			In
Age of regrowth forest		1	8.30667	0.008653			In
Age of regrowth forest	Step 2	1	11.24157	0.002756			In
Nearest forest distance		1	5.40349	0.029283			In
Distance to non-frag. forest		1	3.00102	0.096603			In
Regrowth forest size		1	2.22447	0.149429			In
Cont. regrowth forest size		1	0.78124	0.385909			Removed
Nearest forest size		1			0.088799	0.768506	Out
Age of regrowth forest	Step 3	1	12.61488	0.001621			In
Nearest forest distance		1	4.79517	0.038496			In
Distance to non-frag. forest		1	4.78935	0.038604			In
Regrowth forest size		1	2.45118	0.130528			Removed
Cont. regrowth forest size		1			0.781238	0.385909	Out
Nearest forest size		1			0.001901	0.965603	Out
Age of regrowth forest	Step 4	1	10.87705	0.002919			In
Nearest forest distance		1	2.21928	0.148806			Removed
Distance to non-frag. forest		1	8.07989	0.008786			In
Regrowth forest size		1			2.451184	0.130528	Out
Cont. regrowth forest size		1			0.937764	0.342512	Out
Nearest forest size		1			0.008097	0.929048	Out
Age of regrowth forest	Step 5	1	8.94043	0.006029			In
Distance to non-frag. forest		1	6.17714	0.019698			In
Nearest forest distance		1			2.219280	0.148806	Out
Regrowth forest size		1			0.003576	0.952789	Out
Cont. regrowth forest size		1			0.115278	0.737050	Out
Nearest forest size		1			0.009971	0.921255	Out

Table A.11 Stepwise backward general GRM results for the effects of the independent variables distance to non-fragmented forest and age of regrowth forest *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	2	2808.353	1404.177	15.29548	0.000041	
Residual	26	2386.888	91.80339			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	<i>-95.00%</i>	<i>+95.00%</i>
Intercept	36.51866	5.343806	6.83383	0.000000	25.53431	47.50301
Age	0.73717	0.246540	2.99006	0.006029	0.23040	1.24394
Distance to non-frag. forest	-0.01231	0.004955	-2.48538	0.019698	-0.02250	-0.00213

$$R^2_{adj} = 0.505221$$

$$n = 29$$

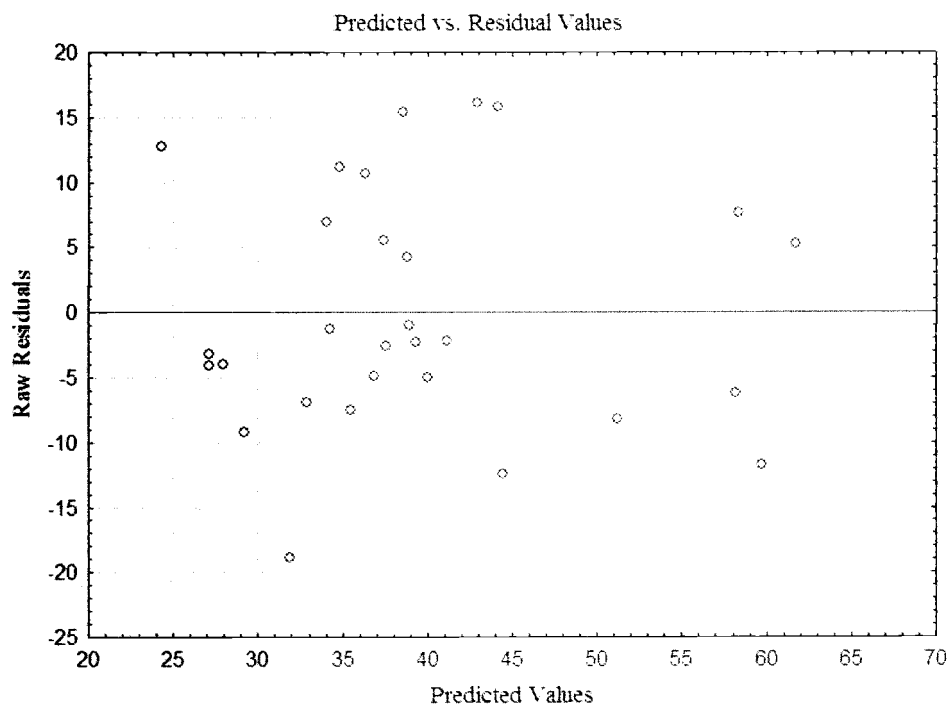


Figure A.18 Predicted *versus* Residuals plot showing a normally distributed residual variances for the independent variables regrowth forest age and distance to non-fragmented forest *versus* the dependent variable species richness.

Table A.12 Multiple linear regression model for the effects of the independent variables $\log_{(10)}$ distance to non-fragmented forest and if burning was used *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	2	2567.092	1283.546	12.69798	0.000142	
Residual	26	2628.149	101.0827			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	87.7450	10.08635	8.69938	0.000000	67.0122	108.4778
$\log_{(10)}$ Distance to non-frag. forest	-19.6841	3.90661	-5.03867	0.000030	-27.7143	-11.6540
Burned	2.4736	3.87100	0.63901	0.528406	-5.4833	10.4306

$$R^2_{adj} = 0.455210$$

$$n = 29$$

Table A.13 Multiple linear regression model for the effects of the independent variables distance to non-fragmented forest, age of regrowth forest and if burning was used in tenure (indicator variable) *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	3	2808.353	949.5542	10.11637	0.000152	
Residual	25	2346.579	93.86315			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	33.07849	7.533602	4.39079	0.000181	17.56275	48.59423
Age	-0.01096	0.005420	-2.02200	0.054001	-0.02212	0.00020
Distance to non-frag. forest	0.81657	0.277174	2.94605	0.006871	0.24572	1.38742
Burned	2.74167	4.183696	0.65532	0.518245	-5.87481	11.35815

$$R^2_{adj} = 0.494120$$

$$n = 29$$

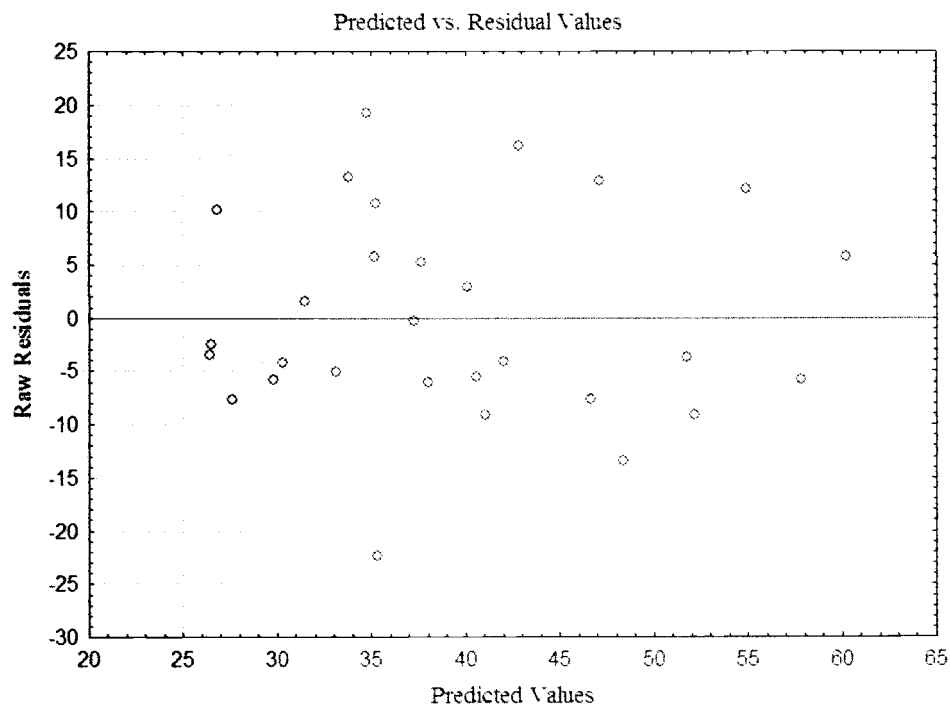


Figure A.19 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables $\log_{(10)}$ distance to non-fragmented forest and if the regrowth forest was burned *versus* the dependent variable species richness.

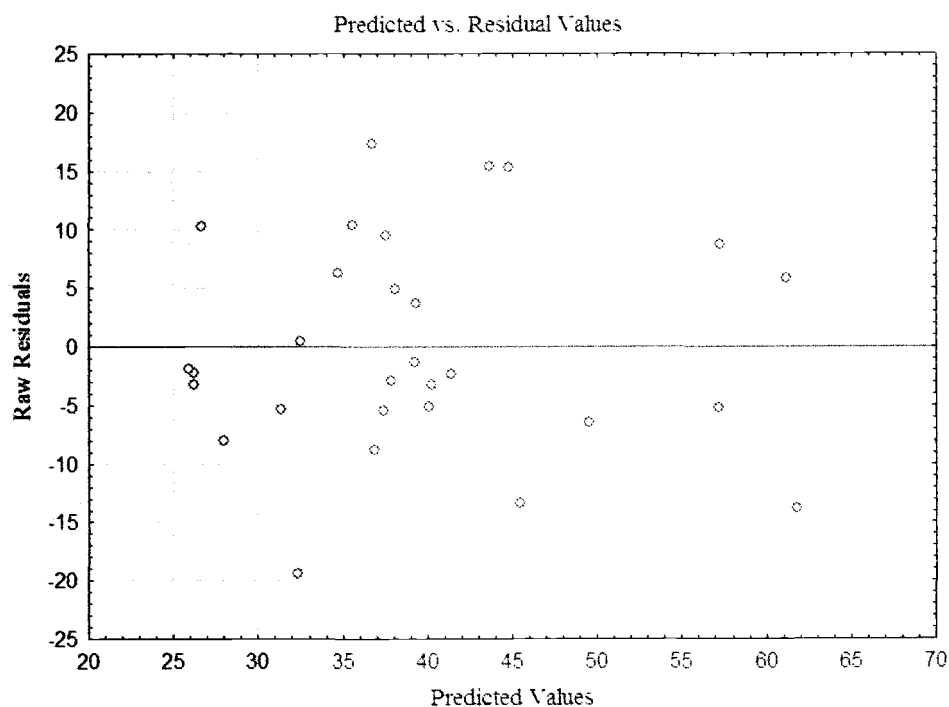


Figure A.20 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables regrowth forest age, distance to non-fragmented forest and if the regrowth forest was burned *versus* the dependent variable species richness.

Table A.14 Results of a simple linear regression model for the effects of the independent variables ArcSine percent of forest within 400 m of the regrowth forest transect *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	1352.990	1352.990	9.507639	0.004678	
Residual	27	3842.251	142.3056			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	31.64193	3.372435	9.382516	0.000000	24.72226	38.56160
Arcsin of % Forest w/in 400 m	25.79914	8.366984	3.083446	0.004678	8.63151	42.96678

$$R^2_{adj} = 0.233037$$

$$n = 29$$

Table A.15 Results of a simple linear regression model for the effects of the independent variables ArcSine percent of fragmented forest within 400 m of the regrowth forest transect *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	732.8122	732.8122	4.433892	0.044671	
Residual	27	4462.429	165.2752			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	42.2127	2.71662	15.53871	0.000000	36.6387	47.78679
Arcsin of % fragmented forest w/in 400 m	-43.6631	20.73586	-2.10568	0.044671	-86.2096	-1.11663

$$R^2_{adj} = 0.109242$$

$$n = 29$$

Table A.16 Results of a simple linear regression model for the effects of the independent variables ArcSine percent of non-fragmented forest within 400 m of the regrowth forest transect *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	2062.091	2062.091	17.77012	0.000250	
Residual	27	3133.151	116.0426			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	32.31558	2.625299	12.30930	0.000000	26.92891	37.70224
Arcsin of % non-frag. forest w/in 400 m	29.77385	7.063011	4.21546	0.000250	15.28175	44.26595

$$R^2_{adj} = 0.374583$$

$$n = 29$$

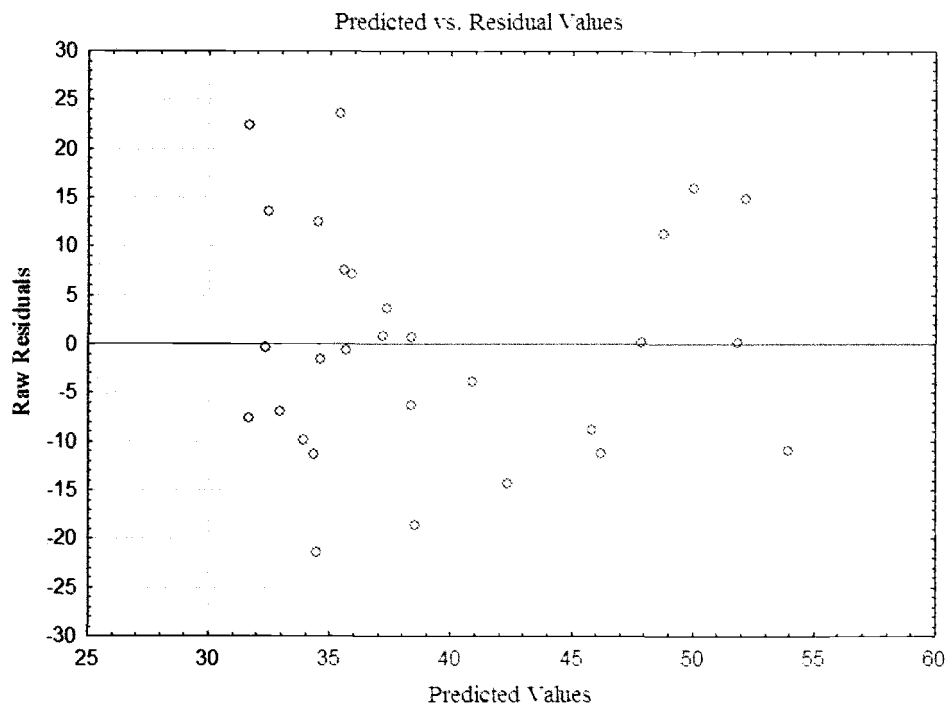


Figure A.21 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables ArcSine percent forest within 400 m of the regrowth transect *versus* the dependent variable species richness.

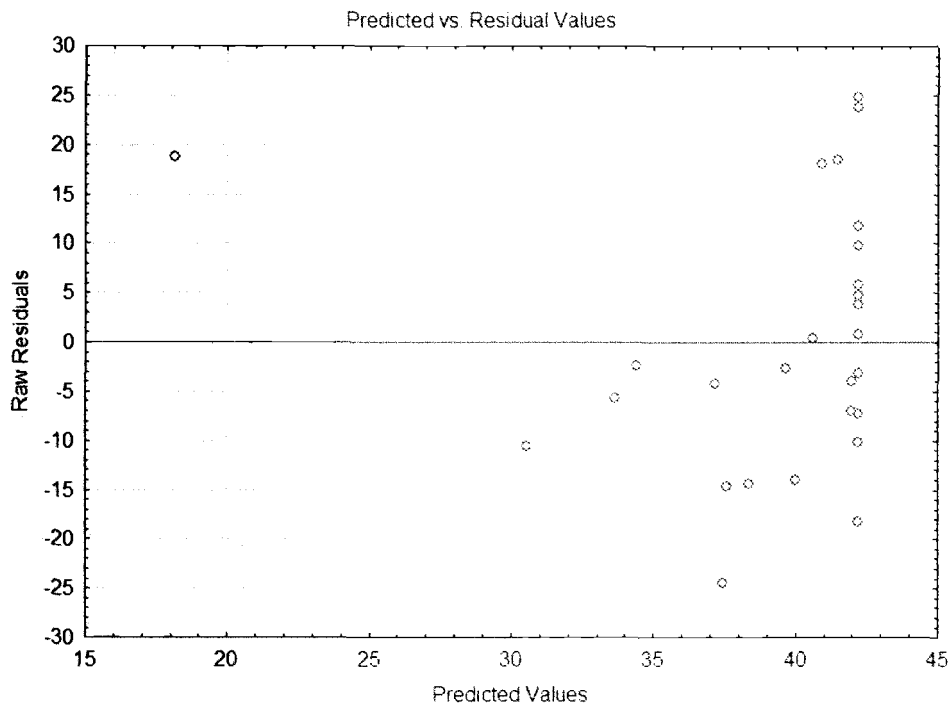


Figure A.22 Predicted *versus* residuals plot showing a non-normally distributed residual variances for the independent variables ArcSine percent fragmented forest within 400 m of the regrowth transect *versus* the dependent variable species richness.

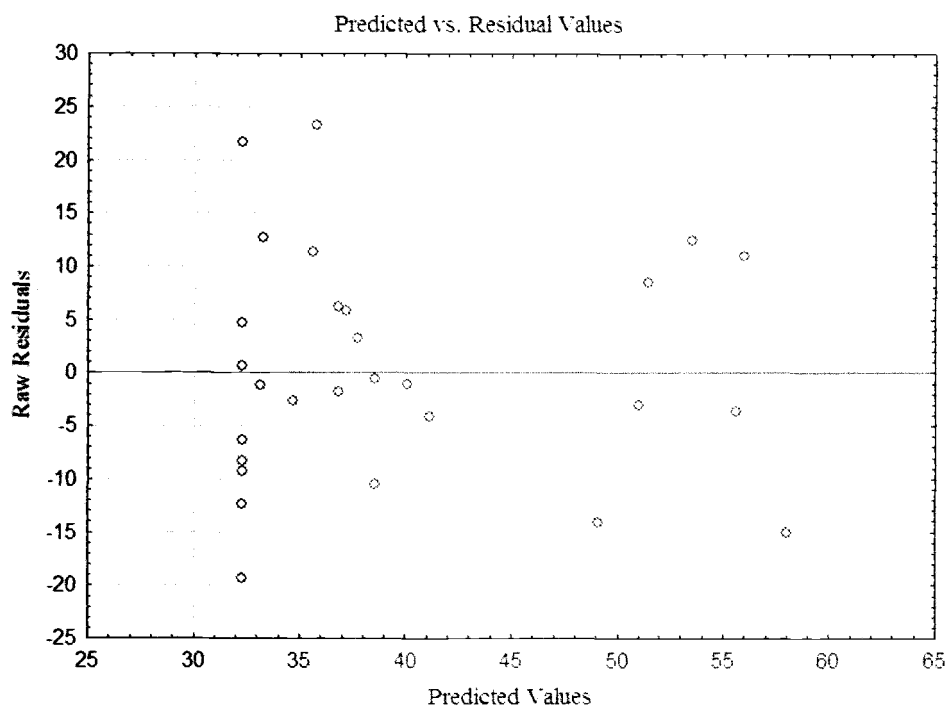


Figure A.23 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables ArcSine percent non-fragmented forest within 400 m of the regrowth transect *versus* the dependent variable species richness.

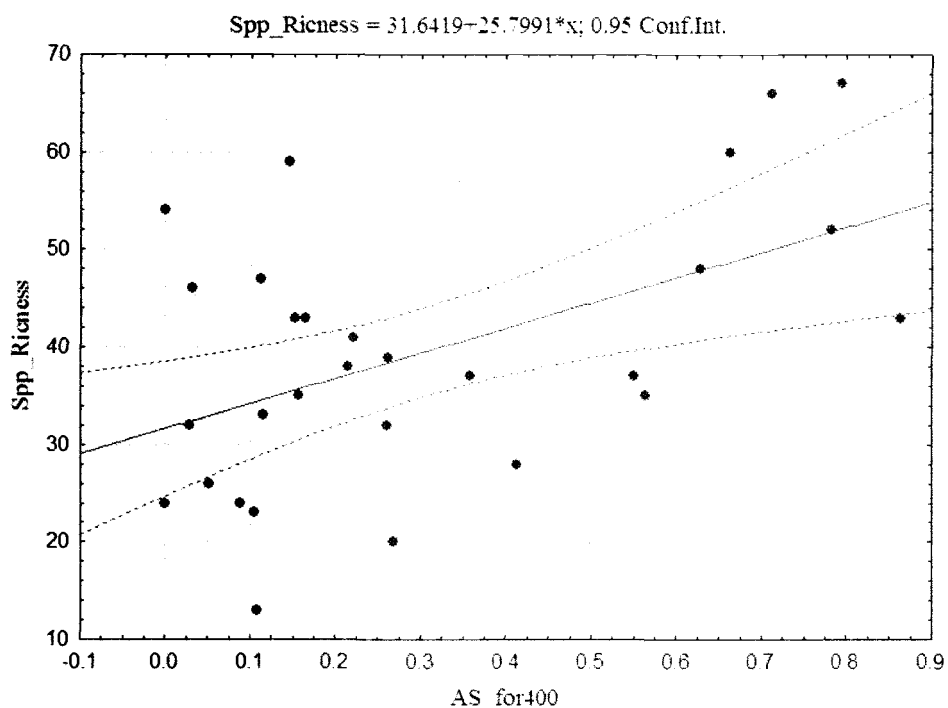


Figure A.24 Scatterplot for ArcSine percent forest within 400 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

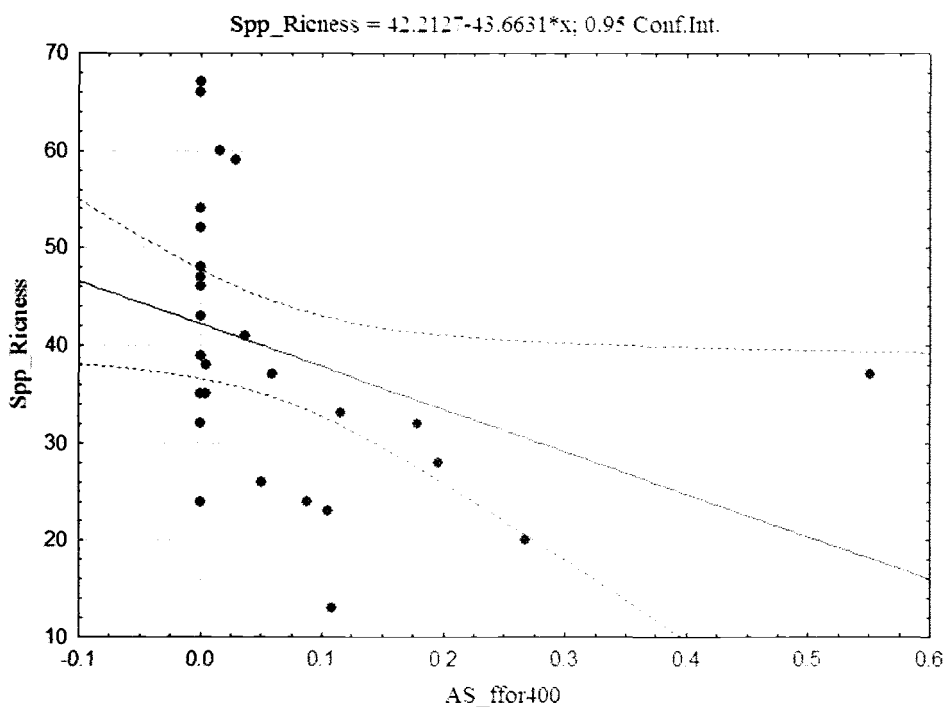


Figure A.25 Scatterplot for ArcSine percent fragmented forest within 400 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

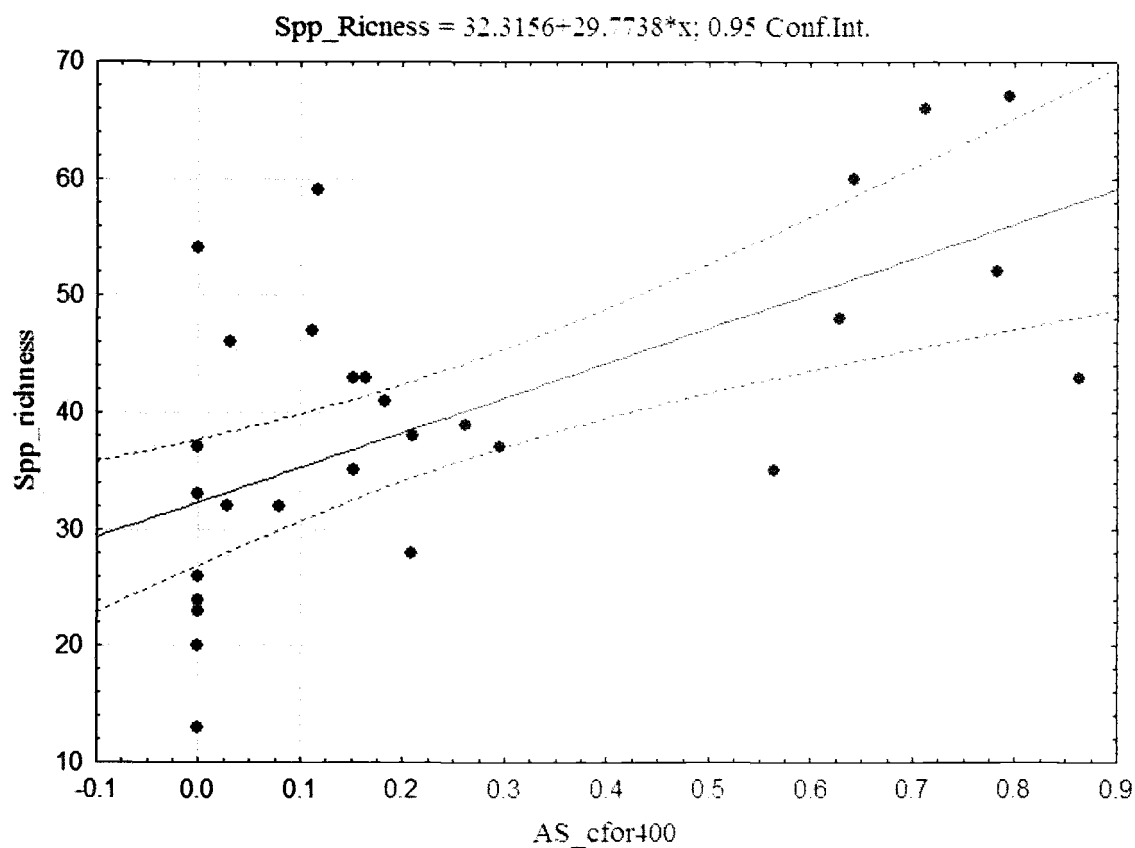


Figure A.26 Scatterplot for ArcSine percent non-fragmented forest within 400 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

Table A.17 Results of a simple linear regression model for the effects of the independent variables ArcSine percent of forest within 800 m of the regrowth forest transect *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	1434.705	1434.705	10.30093	0.003417	
Residual	27	3760.537	139.2791			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	28.70153	4.010816	7.156035	0.000000	20.47202	36.93105
Arcsin of % Forest w/in 800 m	23.96592	7.467167	3.209506	0.003417	8.64456	39.28728

$$R^2_{adj} = 0.249348$$

$$n = 29$$

Table A.18 Results of a simple linear regression model for the effects of the independent variables ArcSine percent of fragmented forest within 800 m of the regrowth forest transect *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	638.1024	638.1024	3.780610	0.062336	
Residual	27	4557.139	168.7829			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	42.7202	2.93127	14.57394	0.000000	36.7057	48.73464
Arcsin of % frag. forest w/in 800 m	-35.6432	18.33142	-1.94438	0.062336	-73.2562	1.96974

$$R^2_{adj} = 0.090336$$

$$n = 29$$

Table A.19 Results of a simple linear regression model for the effects of the independent variables ArcSine percent of non-fragmented forest within 800 m of the regrowth forest transect *versus* the dependent variable species richness.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	1772.038	1772.038	13.97668	0.000880	
Residual	27	3423.203	126.7853			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	31.30860	3.025310	10.34889	0.000000	25.10118	37.51603
Arcsin of % non- frag. forest w/in 800 m	22.91023	6.128123	3.73854	0.000880	10.33636	35.48410

$$R^2_{adj} = 0.316685$$

$$n = 29$$

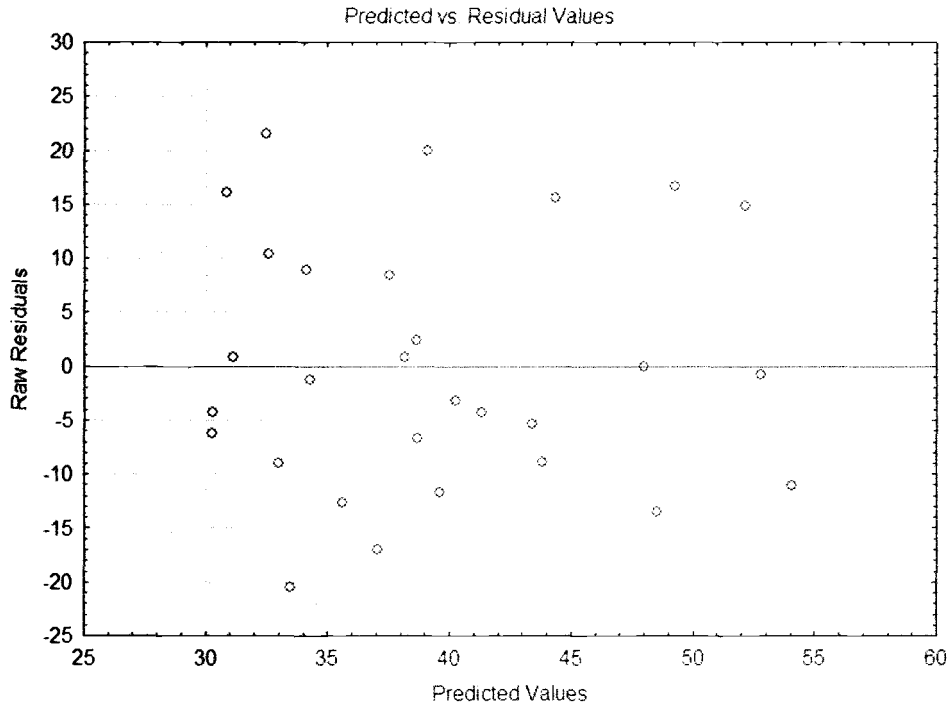


Figure A.27 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables ArcSine percent forest within 800 m of the regrowth transect *versus* the dependent variable species richness.

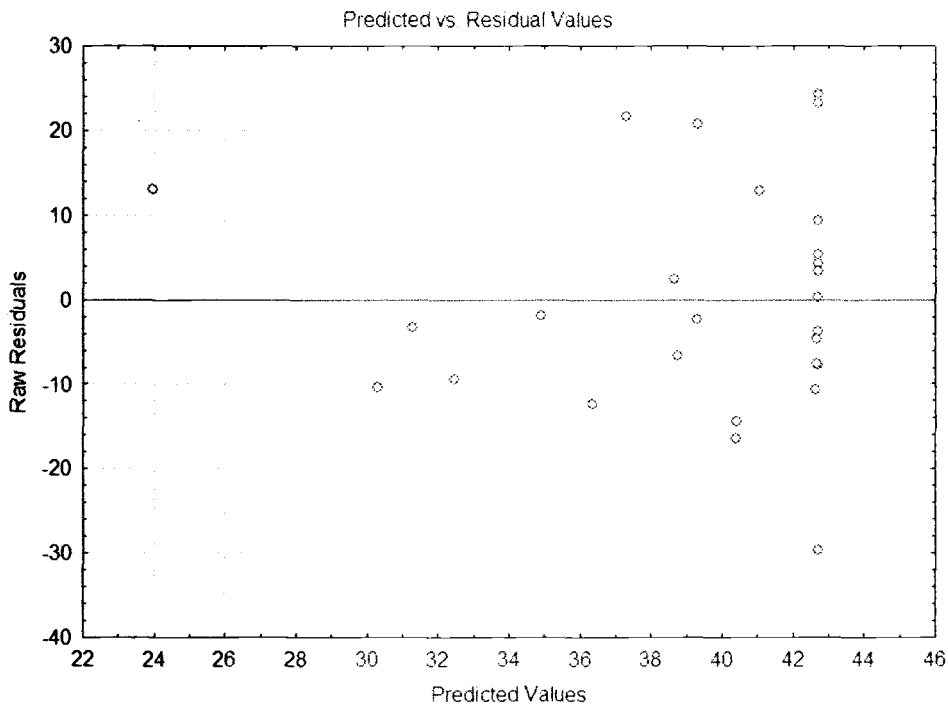


Figure A.28 Predicted *versus* residuals plot showing a non-normally distributed residual variances for the independent variables ArcSine percent fragmented forest within 800 m of the regrowth transect *versus* the dependent variable species richness.

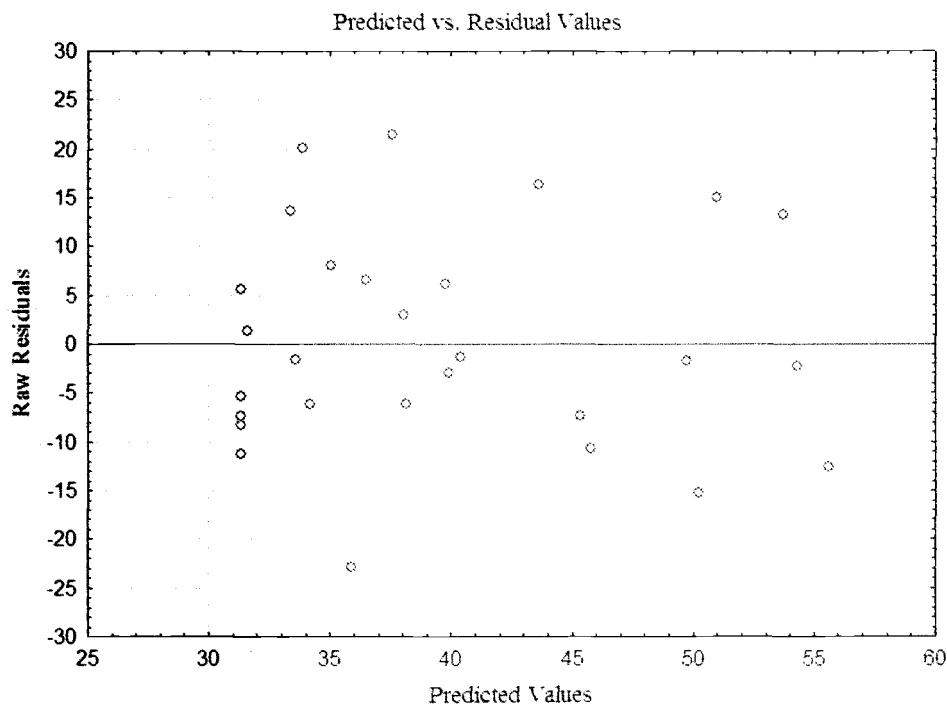


Figure A.29 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables ArcSine percent non-fragmented forest within 800 m of the regrowth transect *versus* the dependent variable species richness.

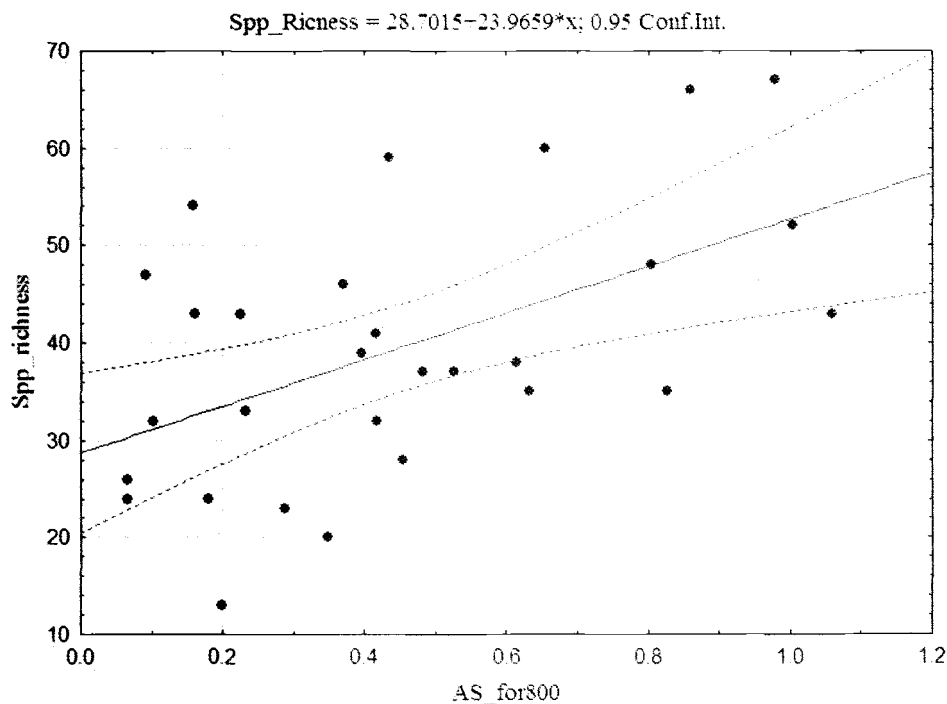


Figure A.30 Scatterplot for ArcSine percent of forest within 800 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

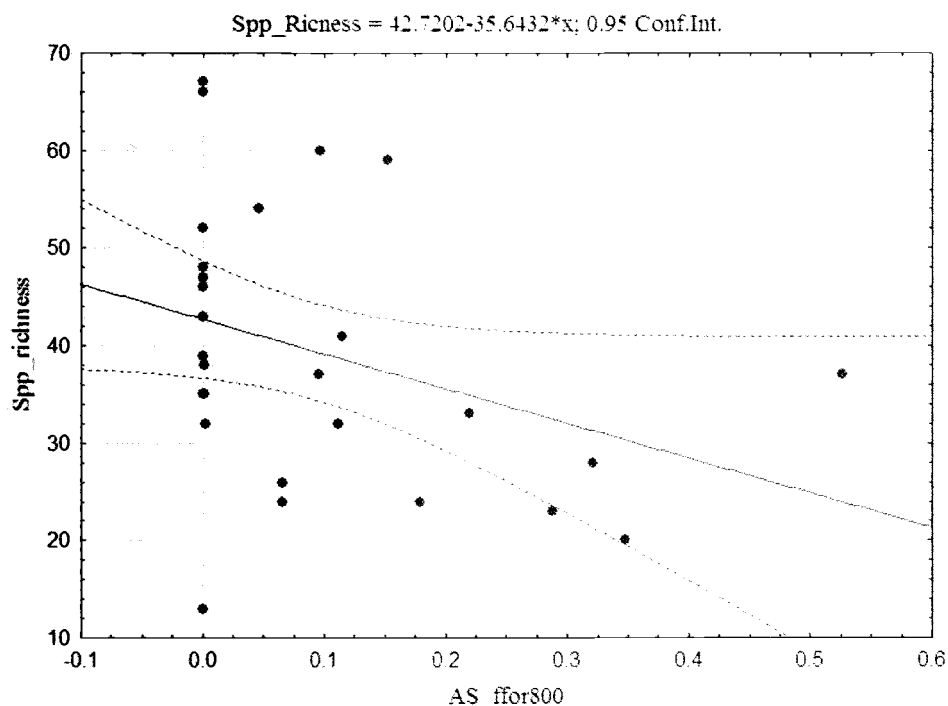


Figure A.31 Scatterplot for ArcSine percent of fragmented forest within 800 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

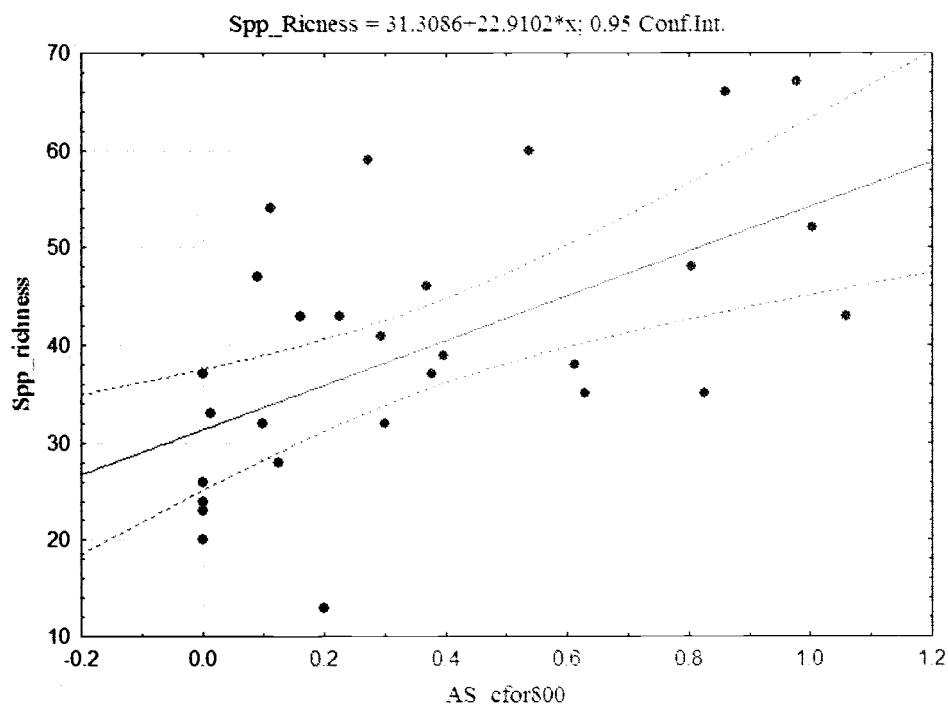


Figure A.32 Scatterplot for ArcSine percent of non-fragmented forest within 800 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line).

Table A.20 Results of a simple linear regression model for the effects of the independent variables percent of forest within 400 m of the regrowth forest transect *versus* the dependent variable species richness after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	1560.580	1560.580	12.93150	0.001699	
Residual	21	2534.290	120.6805			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	31.37492	3.587655	8.745244	0.000000	23.91398	38.83586
% Forest w/in 400 m	33.26155	9.249495	3.596039	0.001699	14.02617	52.49693

$$R^2_{adj} = 0.351635$$

$$n = 23$$

Table A.21 Results of a simple linear regression model for the effects of the independent variables percent of fragmented forest within 400 m of the regrowth forest transect *versus* the dependent variable species richness after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	1674.873	1674.873	14.53405	0.001017	
Residual	21	2419.996	115.2379			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	47.138	2.71149	17.38470	0.000000	41.500	52.7772
% Fragmented forest w/in 400 m	-216.601	56.81543	-3.81235	0.001017	-334.755	-98.4464

$$R^2_{adj} = 0.380875$$

$$n = 23$$

Table A.22 Results of a simple linear regression model for the effects of the independent variables percent of non-fragmented forest within 400 m of the regrowth forest transect *versus* the dependent variable species richness after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

ANOVA						
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>p</i>	
Regression	1	1828.672	1828.672	16.94561	0.000492	
Residual	21	2266.198	107.9142			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>p</i>	-95.00%	+95.00%
Intercept	32.21666	3.092817	10.41661	0.000000	25.78479	38.64852
% Non-frag. forest w/in 400 m	33.46092	8.128479	4.11651	0.000492	16.55683	50.36502

$$R^2_{adj} = 0.420223$$

$$n = 23$$

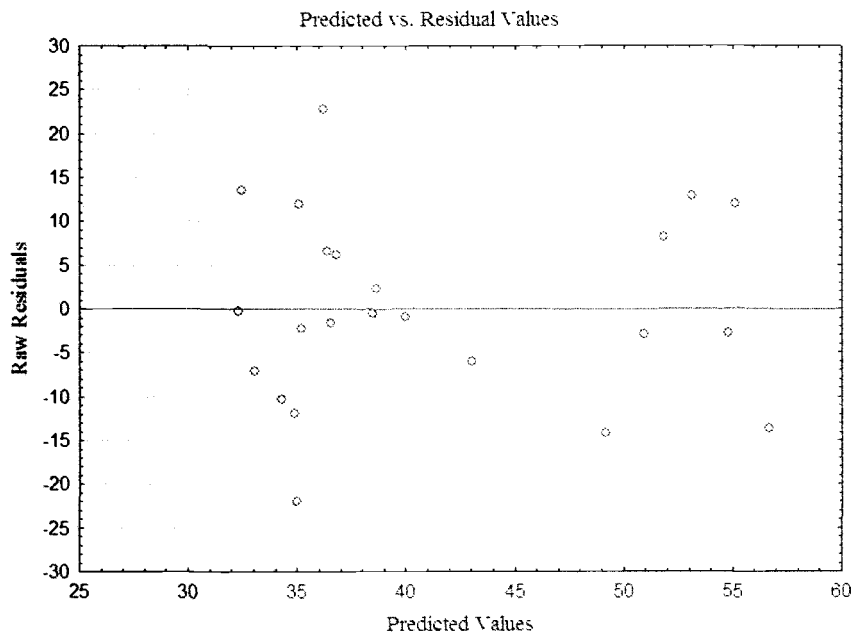


Figure A.33 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables percent forest within 400 m of the regrowth transect *versus* the dependent variable species richness after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

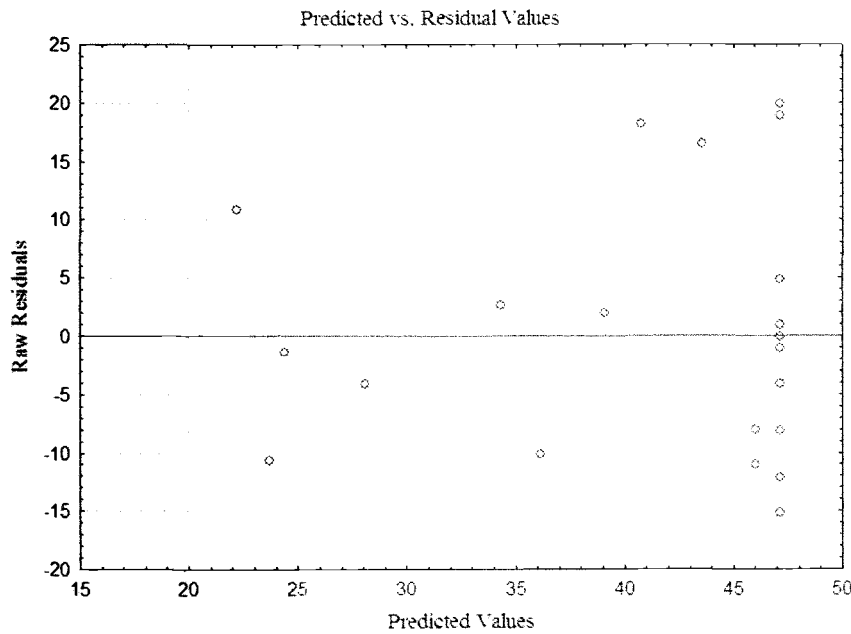


Figure A.34 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables percent fragmented forest within 400 m of the regrowth transect *versus* the dependent variable species richness after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

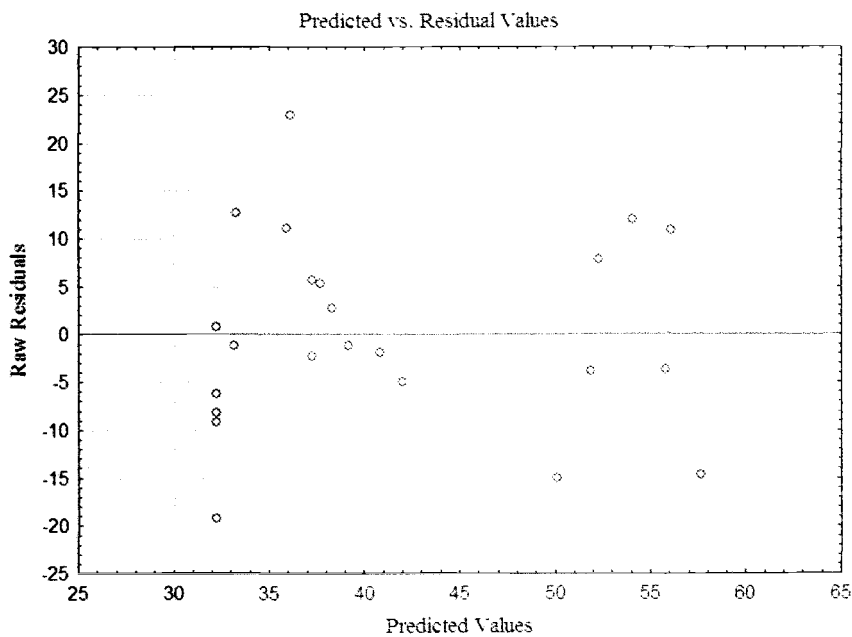


Figure A.35 Predicted *versus* residuals plot showing a normally distributed residual variances for the independent variables percent non-fragmented forest within 400 m of the regrowth transect *versus* the dependent variable species richness after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

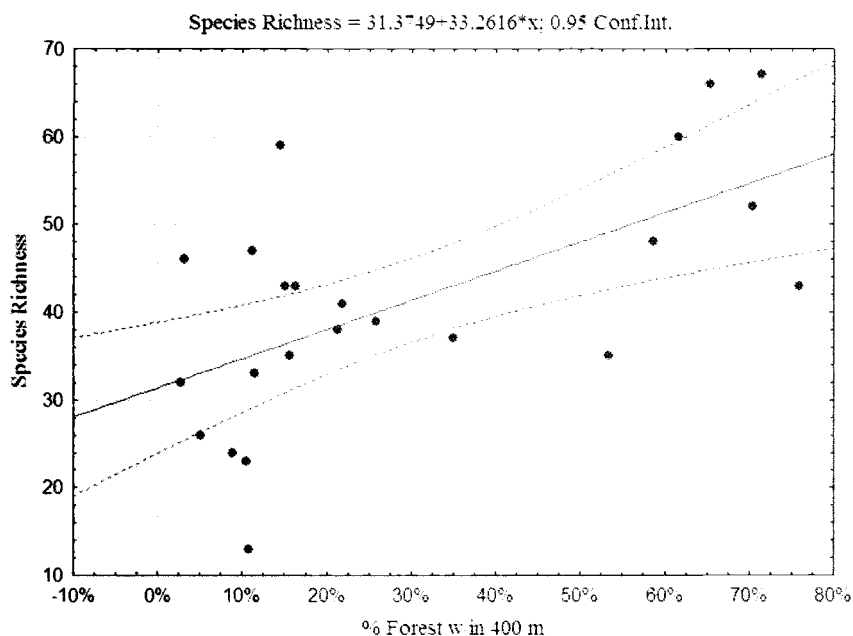


Figure A.36 Scatterplot for percent of forest within 400 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line) after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

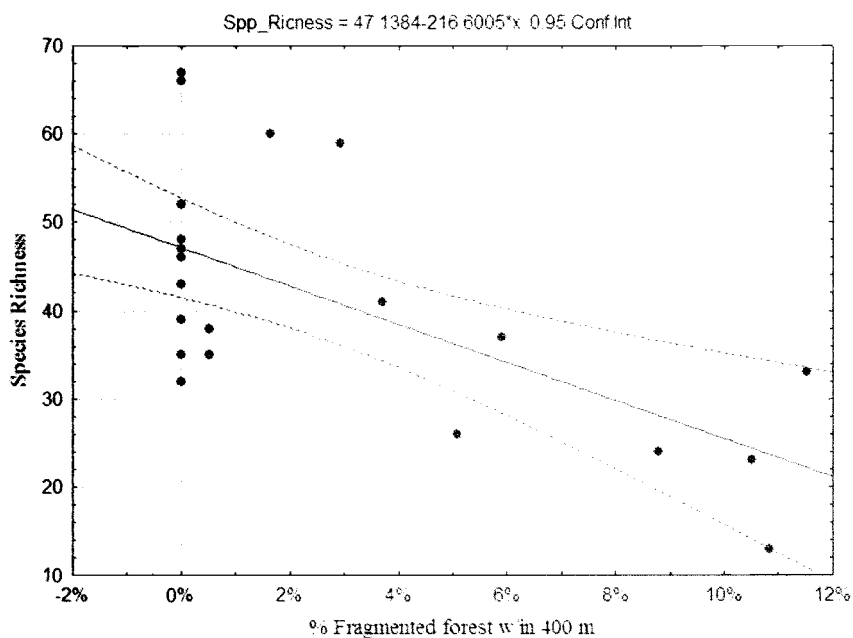


Figure A.37 Scatterplot for percent of fragmented forest within 400 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line) after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

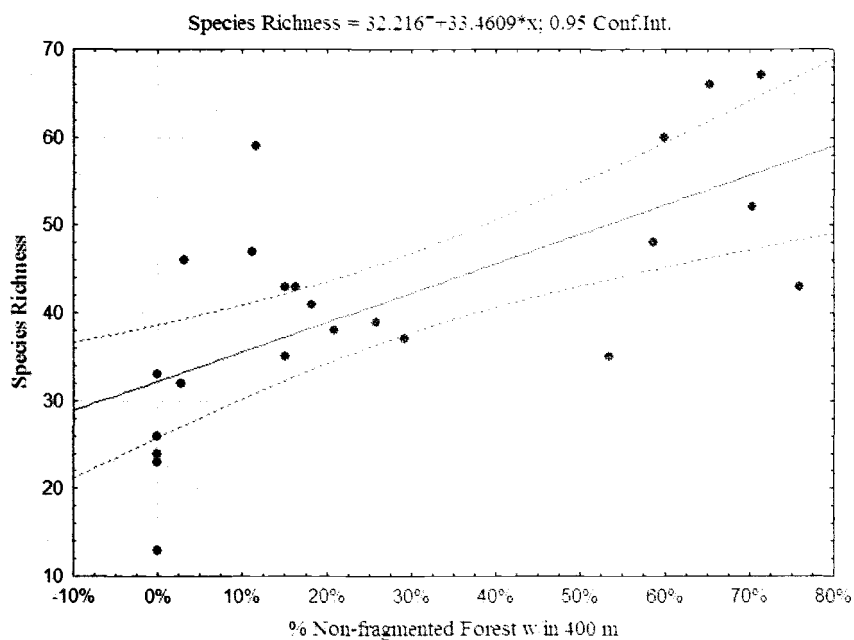


Figure A.38 Scatterplot for percent of non-fragmented forest within 400 m of the regrowth forest transect *versus* species richness, linear regression line (red) with 95% confidence interval (dashed red line) after data with zero's across all forest classes and high leverage cases are removed to meet normality of residual variance.

APPENDIX B LINEAR REGRESSION RESULTS TABLES FOR CHAPTER THREE

Table B.1 Linear Regression results for aboveground biomass for Transects 1.

Total Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.99291274
R Square	0.98587572
Adjusted R Square	0.98478924
Standard Error	44965.2172
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	1.83465E+12	1.8346E+12	907.400812	2.06101E-13
Residual	13	26284319795	2021870753		
Total	14	1.86093E+12			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	73624969.5	24432.21787	3013.43782	2.2359E-39	73572186.87	73677752
Time	80946.3483	2687.185698	30.1230943	2.061E-13	75141.03762	86751.6589

Mature Forest Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.96771205
R Square	0.93646661
Adjusted R Square	0.93157943
Standard Error	53505.2775
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	5.48563E+11	5.4856E+11	191.616816	3.70208E-09
Residual	13	37216591301	2862814715		
Total	14	5.8578E+11			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	73465888.5	29072.52936	2526.98648	2.205E-38	73403081.16	73528695.9
Time	-44262.345	3197.551918	-13.842573	3.7021E-09	-51170.2342	-37354.455

Regrowth Forest Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.99743283
R Square	0.99487226
Adjusted R Square	0.99447781
Standard Error	41668.4379
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	4.37924E+12	4.3792E+12	2522.22783	2.83213E-16
Residual	13	22571363274	1736258713		
Total	14	4.40181E+12			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	161411.378	22640.88592	7.12919884	7.7108E-06	112498.7269	210324.029
Time	125060.557	2490.165452	50.2217864	2.8321E-16	119680.8831	130440.232

Table B.2 Linear Regression results for aboveground biomass for Transect 2.

Total Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.99234379
R Square	0.98474619
Adjusted R Square	0.98357282
Standard Error	161383.607
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	2.18579E+13	2.1858E+13	839.246191	3.39996E-13
Residual	13	3.38581E+11	2.6045E+10		
Total	14	2.21965E+13			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	73757265.2	87689.10078	841.122381	3.5813E-32	73567824.48	73946706
Time	-279399.16	9644.515236	-28.969746	3.4E-13	-300234.862	-258563.45

Mature Forest Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.99474601
R Square	0.98951961
Adjusted R Square	0.98871343
Standard Error	162292.954
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	3.23288E+13	3.2329E+13	1227.41241	2.95841E-14
Residual	13	3.42407E+11	2.6339E+10		
Total	14	3.26712E+13			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	73409836.1	88183.20164	832.469617	4.0965E-32	73219327.89	73600344.3
Time	-339794.16	9698.859085	-35.034446	2.9584E-14	-360747.262	-318841.05

Regrowth Forest Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.9875579
R Square	0.9752706
Adjusted R Square	0.97336834
Standard Error	44546.2862
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	1.01737E+12	1.0174E+12	512.690111	7.89258E-12
Residual	13	25796831040	1984371618		
Total	14	1.04316E+12			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	357350.825	24204.58834	14.7637638	1.6789E-09	305060.0011	409641.649
Time	60278.156	2662.149788	22.6426613	7.8926E-12	54526.93212	66029.3798

Table B.3 Linear Regression results for aboveground biomass for Transect 3.

Total Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.978421536
R Square	0.957308703
Adjusted R Square	0.953751095
Standard Error	635335.6986
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	1.08618E+14	1.08618E+14	269.0877333	1.39139E-09
Residual	12	4.84382E+12	4.03651E+11		
Total	13	1.13461E+14			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	41259000.15	358658.8579	115.0368916	1.24785E-19	40477549.64	42040450.66
Time	-690970.503	42122.34665	-16.4038938	1.39139E-09	-782747.2107	-599193.795

Mature Forest Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.989580171
R Square	0.979268916
Adjusted R Square	0.977541325
Standard Error	500292.8579
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	1.41876E+14	1.41876E+14	566.840925	1.80692E-11
Residual	12	3.00352E+12	2.50293E+11		
Total	13	1.4488E+14			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	40352000.94	282424.6543	142.8770482	9.27744E-21	39736650.49	40967351.39
Time	-789703.752	33169.0935	-23.8084213	1.80692E-11	-861972.9975	-717434.507

Regrowth Forest Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.94065551
R Square	0.88483279
Adjusted R Square	0.87523552
Standard Error	152373.124
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	2.14057E+12	2.1406E+12	92.1963245	5.5449E-07
Residual	12	2.78611E+11	2.3218E+10		
Total	13	2.41919E+12			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	938602.137	86017.47192	10.9117615	1.3843E-07	751186.169	1126018.11
Time	97000.6121	10102.23975	9.60189171	5.5449E-07	74989.7229	119011.501

Table B.4 Linear Regression results for aboveground biomass for combined transects.

Total Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.98667507
R Square	0.9735277
Adjusted R Square	0.97132168
Standard Error	634694.003
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	1.77773E+14	1.7777E+14	441.304041	7.85322E-11
Residual	12	4.83404E+12	4.0284E+11		
Total	13	1.82607E+14			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	188414447	358296.6088	525.861654	1.506E-27	187633786	189195108
Time	-883980.42	42079.80266	-21.007238	7.8532E-11	-975664.435	-792296.41

Mature Forest Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.9951635
R Square	0.9903504
Adjusted R Square	0.98954627
Standard Error	501219.374
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	3.09397E+14	3.094E+14	1231.57467	1.82882E-13
Residual	12	3.01465E+12	2.5122E+11		
Total	13	3.12412E+14			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	186770444	282947.69	660.088243	9.8426E-29	186153953.5	187386934
Time	-1166185.2	33230.52092	-35.093798	1.8288E-13	-1238588.28	-1093782.1

Regrowth Forest Aboveground Biomass

<i>Regression Statistics</i>	
Multiple R	0.990749831
R Square	0.981585227
Adjusted R Sq	0.980050663
Standard Error	168299.8041
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	1.8118E+13	1.8118E+13	639.650714	8.86678E-12
Residual	12	3.39898E+11	28324824070		
Total	13	1.84579E+13			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	1644003.73	95008.38008	17.30377603	7.50952E-10	1436998.256	1851009.204
Time	282204.7728	11158.16836	25.29131697	8.86678E-12	257893.2128	306516.3328

Table B.5 Linear regression results for time versus average regrowth forest age for Transect 1, Transect 2, Transect 3 and combined transects for the years of 1986 to 1999.

Transect 1

<i>Regression Statistics</i>	
Multiple R	0.99886255
R Square	0.99772639
Adjusted R Square	0.9975515
Standard Error	0.1580013
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	142.4163799	142.41638	5704.77639	1.43123E-18
Residual	13	0.324537337	0.02496441		
Total	14	142.7409172			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	2.32139934	0.085851296	27.0397705	8.2183E-13	2.135928931	2.50686976
Time	0.71318296	0.009442384	75.5299701	1.4312E-18	0.69278393	0.73358198

Transect 2

<i>Regression Statistics</i>	
Multiple R	0.92832045
R Square	0.86177885
Adjusted R Square	0.85114646
Standard Error	0.28553198
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	6.608063324	6.60806332	81.0521789	5.99956E-07
Residual	13	1.059870621	0.08152851		
Total	14	7.667933945			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	3.35852375	0.15514613	21.6474864	1.3959E-11	3.023350973	3.69369652
Time	0.15362365	0.017063799	9.00289836	5.9996E-07	0.116759561	0.19048774

Transect 3

<i>Regression Statistics</i>	
Multiple R	0.78738513
R Square	0.61997534
Adjusted R Square	0.58830662
Standard Error	0.32565194
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	2.076114106	2.07611411	19.5768982	0.000828916
Residual	12	1.272590221	0.10604919		
Total	13	3.348704327			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	2.61698163	0.183836596	14.2353681	7.0589E-09	2.216436099	3.01752715
Time	0.09552893	0.021590513	4.42457888	0.00082892	0.048487243	0.14257062

Table B.5 Linear regression results for time versus average regrowth forest age for Transect 1, Transect 2, Transect 3 and combined transects for the years of 1986 to 1999 (Continued).

Combined Transects

<i>Regression Statistics</i>	
Multiple R	0.96243532
R Square	0.92628174
Adjusted R Square	0.92013855
Standard Error	0.24057874
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	8.72697501	8.72697501	150.781914	3.74069E-08
Residual	12	0.694537545	0.05787813		
Total	13	9.421512556			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	2.9428919	0.135811187	21.6689948	5.4609E-11	2.646984747	3.23879905
Time	0.19585793	0.015950215	12.2793287	3.7407E-08	0.1611054	0.23061046

Table B.6 Linear regression results for time versus area deforested for Transect 1, Transect 2, Transect 3 and combined transects for the years of 1986 to 1999.

Transect 1

<i>Regression Statistics</i>	
Multiple R	0.96771205
R Square	0.93646661
Adjusted R Square	0.93157943
Standard Error	150.295723
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	4328394.782	4328394.78	191.616816	3.70208E-09
Residual	13	293654.4573	22588.8044		
Total	14	4622049.24			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	19975.5146	81.66440833	244.604902	3.3623E-25	19799.08938	20151.9398
Time	124.332429	8.981887411	13.8425726	3.7021E-09	104.9282443	143.736613

Transect 2

<i>Regression Statistics</i>	
Multiple R	0.99474601
R Square	0.98951961
Adjusted R Square	0.98871343
Standard Error	455.879084
Observations	15

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	255087891.6	255087892	1227.41241	2.95841E-14
Residual	13	2701734.613	207825.739		
Total	14	257789626.2			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	20206.2253	247.7056226	81.5735431	5.272E-19	19671.08993	20741.3606
Time	954.477964	27.24398619	35.034446	2.9584E-14	895.6209218	1013.33501

Transect 3

<i>Regression Statistics</i>	
Multiple R	0.98958017
R Square	0.97926892
Adjusted R Square	0.97754133
Standard Error	1405.31702
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	1119463165	1119463165	566.840925	1.80692E-11
Residual	12	23698991.01	1974915.92		
Total	13	1143162156			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	64410.7727	793.3276806	81.1906282	8.127E-18	62682.26025	66139.2852
Time	2218.26897	93.17161095	23.8084213	1.8069E-11	2015.265469	2421.27246

Table B.6 Linear regression results for time versus area deforested for Transect 1, Transect 2, Transect 3 and combined transects for the years of 1986 to 1999 (Continued).

Combined Transects

<i>Regression Statistics</i>	
Multiple R	0.9951635
R Square	0.9903504
Adjusted R Square	0.98954627
Standard Error	1407.91959
Observations	14

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	2441273587	2441273587	1231.57467	1.82882E-13
Residual	12	23786850.87	1982237.57		
Total	13	2465060438			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	105877.013	794.796882	133.212667	2.1489E-20	104145.2991	107608.726
Time	3275.80111	93.3441599	35.0937982	1.8288E-13	3072.42166	3479.18056

Table B.7 Linear regression results for time versus Carbon emissions for Transect 1, Transect 2, Transect 3 and combined transects for the years of 1986 to 1999.

Transect 1

<i>Regression Statistics</i>	
Multiple R	0.32717136
R Square	0.1070411
Adjusted R Square	-0.0045788
Standard Error	6824.82077
Observations	10

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	44667492.42	44667492.4	0.95897894	0.356125886
Residual	8	372625427.9	46578178.5		
Total	9	417292920.3			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	39627.8034	7457.313336	5.31395177	0.00071624	22431.19691	56824.4099
Time	-735.8153	751.3880366	-0.9792747	0.35612589	-2468.52034	996.889744

Transect 2

<i>Regression Statistics</i>	
Multiple R	0.09906554
R Square	0.00981398
Adjusted R Square	-0.1002067
Standard Error	42827.1444
Observations	11

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	163609752.2	163609752	0.08920125	0.771976108
Residual	9	16507478676	1834164297		
Total	10	16671088428			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	160956.436	38953.22965	4.13204342	0.0025517	72838.04158	249074.831
Time	1219.57426	4083.407999	0.29866579	0.77197611	-8017.74343	10456.892

Transect 3

<i>Regression Statistics</i>	
Multiple R	0.57050139
R Square	0.32547184
Adjusted R Square	0.25052426
Standard Error	105382.376
Observations	11

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	48227173488	4.8227E+10	4.34266011	0.066840705
Residual	9	99949006084	1.1105E+10		
Total	10	1.48176E+11			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	596551.327	95850.04889	6.22379784	0.0001544	379723.2872	813379.367
Time	-20938.693	10047.81529	-2.083905	0.06684071	-43668.4473	1791.06199

Table B.7 Linear regression results for time versus Carbon emissions for Transect 1, Transect 2, Transect 3 and combined transects for the years of 1986 to 1999 (Continued).

Combined transects

<i>Regression Statistics</i>	
Multiple R	0.47906554
R Square	0.22950379
Adjusted R Square	0.13319177
Standard Error	116724.451
Observations	10

ANOVA

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	1	32466317468	3.2466E+10	2.38291938	0.161244955
Residual	8	1.08997E+11	1.3625E+10		
Total	9	1.41463E+11			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	770507.485	115300.929	6.68257829	0.00015548	504622.8935	1036392.08
Time	-19837.619	12850.93913	-1.5436707	0.16124496	-49471.95676	9796.7191

APPENDIX C PRESENT INFRASTRUCTURE, DEVELOPMENT PLANS AND SPATIAL DATA OF THE BRAZILIAN LEGAL AMAZON

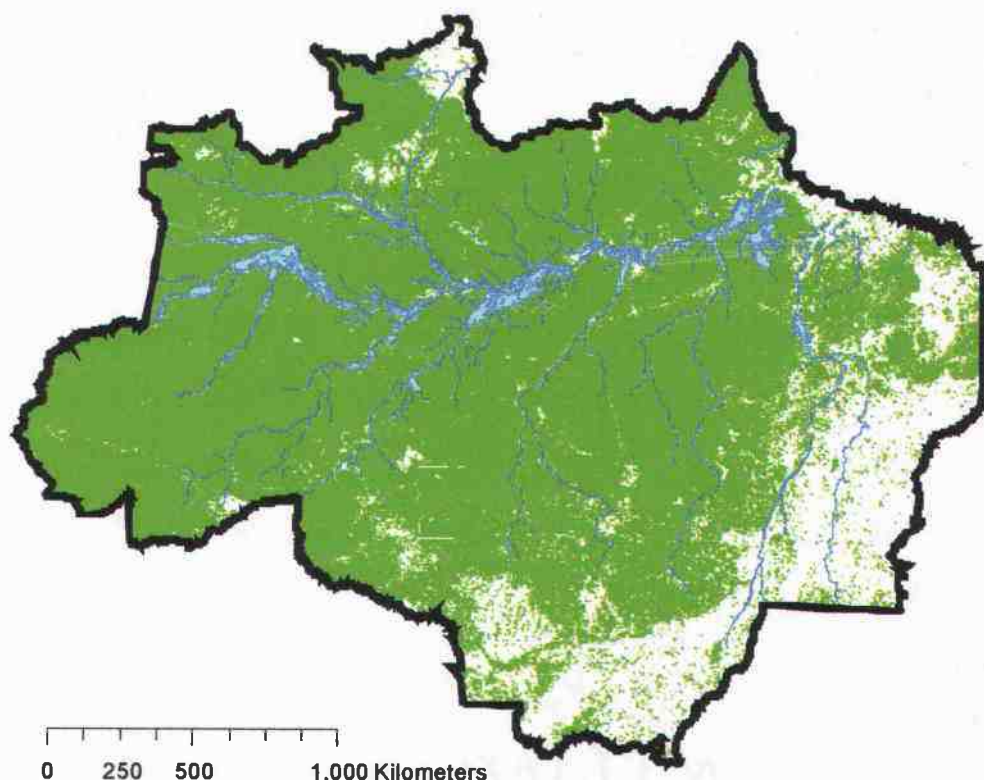


Figure C.1 Spatial extent of forest occurring within the Brazilian Legal Amazon.

Source: Forest/nonforest coverage produced by the National Oceanographic and Atmospheric Administration based on 1999 AVHRR imagery with areas that were illuminated by light sources in the 1999- 2000 year by The Defense Meteorological Satellite Program (DMSP) were assumed to be inhabited/ nonforested due to fire or settlement (<http://dmisp.ngdc.noaa.gov/>).

Total Area of Forest: 3,690,989 km².

Use in Models: The model only considers forested areas of the Brazilian Legal Amazon. Non-forested areas are automatically categorized as heavily degraded. Forested areas are classified as heavily degraded, moderately degraded, lightly degraded or pristine.

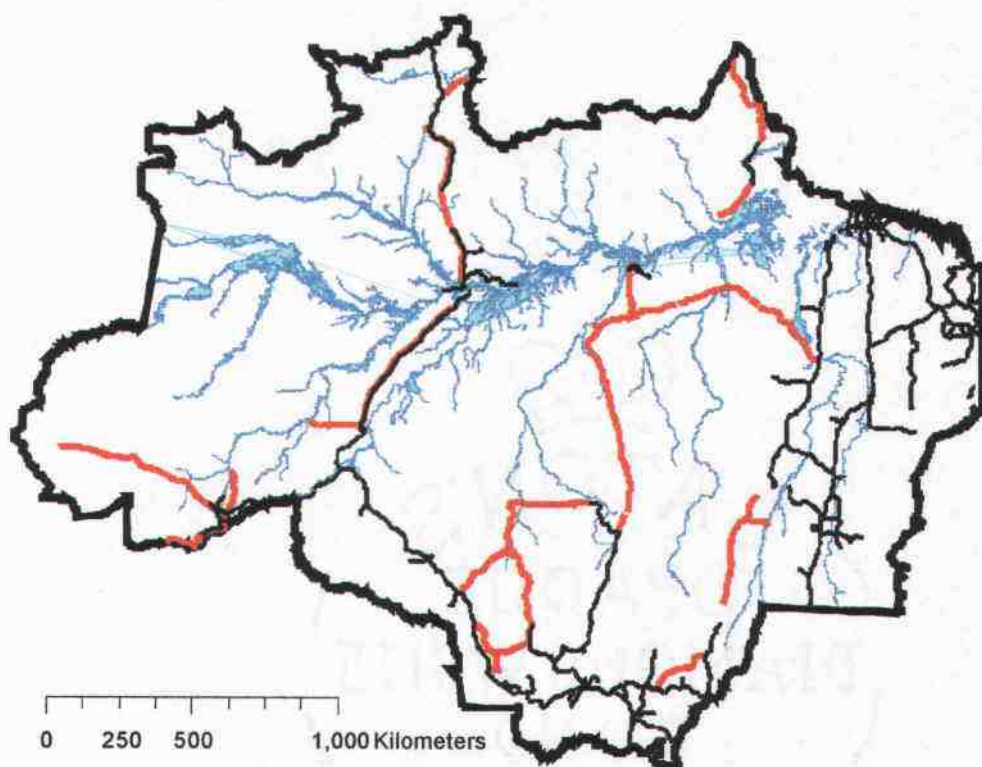


Figure C.2 Spatial extent of current (black line) and proposed improved roads (red line) within the Brazilian Legal Amazon (*Avança Brasil*, 2001).

Source: 1995 map of Brazilian Legal Amazon (1:3,000,000 scale) produced by Brazilian Institute for Geography and Statistics (IBGE); supplemented by 1999 map of Amazonian protected areas (1:4,000,000 scale, *Instituto Socio-Ambiental*, São Paulo, Brazil), JERS-1 radar imagery for 1999 (<http://daac.ornl.gov>), and personal knowledge. Information on planned roads were obtained from information provided by *Avança Brasil* (1), *Brasil em Ação* (2), and personal knowledge.

1) *Avança Brasil: Development Structures for Investment* (Ministry for Development, Industry, and Foreign Trade, Brasília, Brazil, 1999)

2) *Identificação de Oportunidades de Investimentos Públicos e/ou Privados: Estudo de Eixos Nacionais de Integração de Desenvolvimento* (Programa Brasil em Ação, Brasília, Brazil, 2000)

Total Paved Roads: 4,329 km

Total Planed Upgrades: 7,284 km

Use in Model: Areas within 75 km of improved roads is classified as lightly degraded, areas within 50 km are moderately degraded and areas within 25 km are heavily degraded.

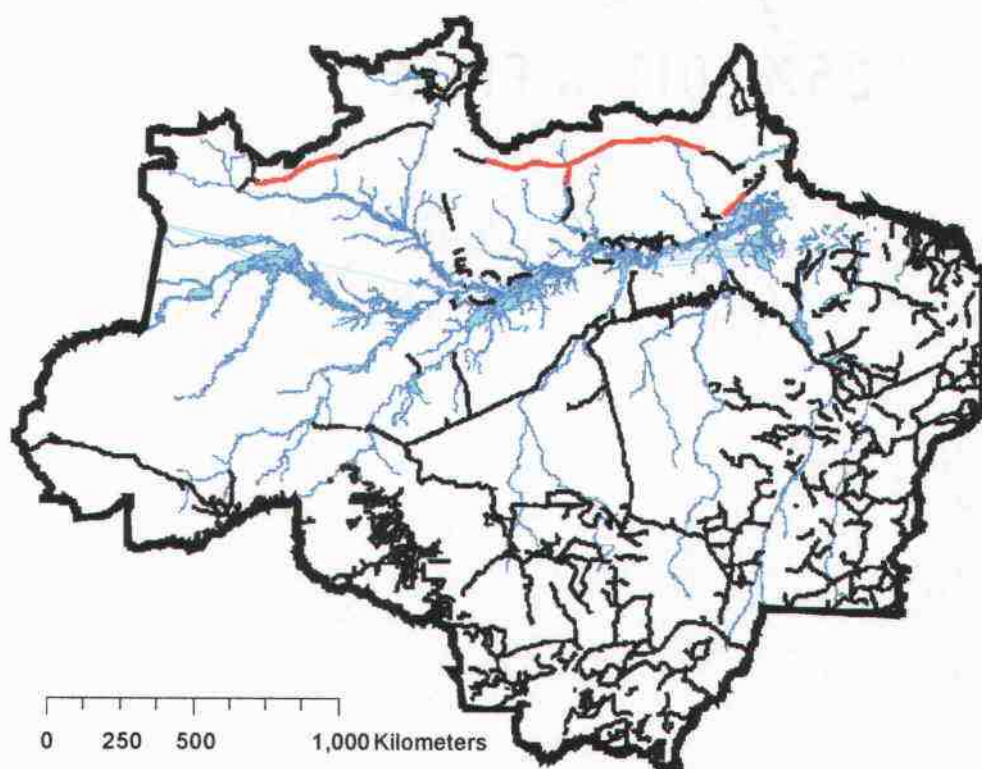


Figure C.3 Spatial distribution of current unpaved roads (black line) and the proposed new road construction (red line) within the Brazilian Legal Amazon.

Source: 1995 map of Brazilian Legal Amazon (1:3,000,000 scale) produced by Brazilian Institute for Geography and Statistics (IBGE); supplemented by 1999 map of Amazonian protected areas (1:4,000,000 scale, *Instituto Socio-Ambiental*, São Paulo, Brazil), JERS-1 radar imagery for 1999 (<http://daac.ornl.gov>), and personal knowledge. Information on planned roads were obtained from information provided by *Avança Brasil* (1), *Brasil em Ação* (2), and personal knowledge.

Total Unpaved Roads:	46,375 km
Total Planned Roads:	1,275 km

Use In Model: Areas within 50 km of an unpaved road are lightly degraded. Areas within 25 km are moderately degraded. Areas within 10 km are heavily degraded.

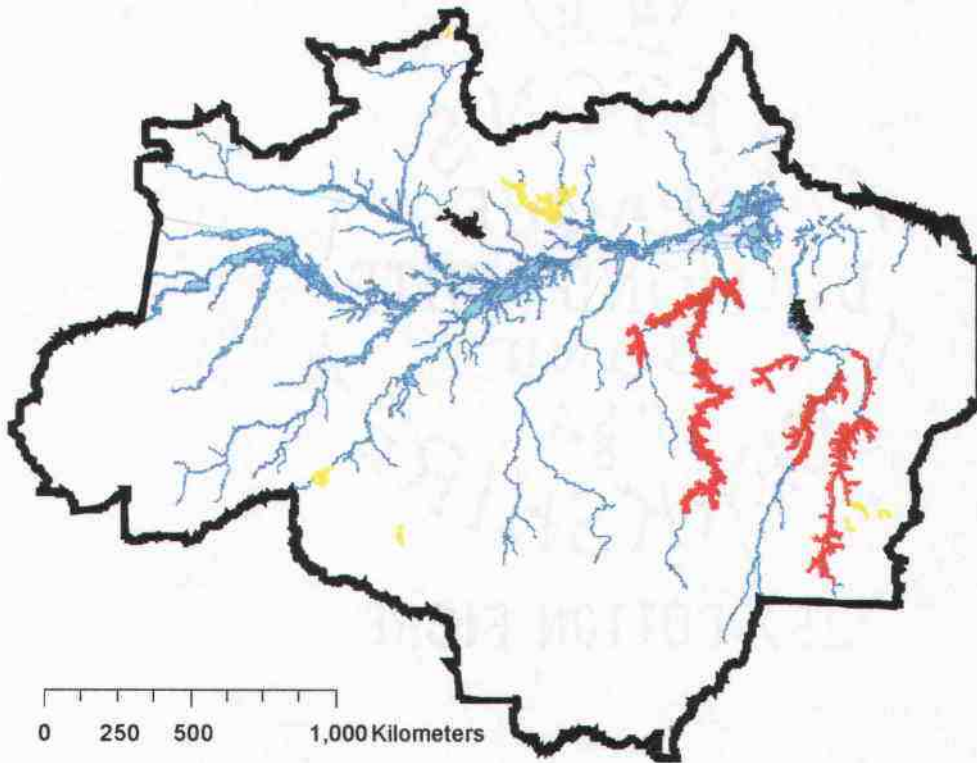


Figure C.4 Spatial extent of current (black), proposed by 2011 (red), and proposed by 2021 (yellow) dams occurring within the Brazil Legal Amazon.

Source: Existing hydroelectric projects were from 1995 IBGE map of Brazilian Legal Amazon (3) and personal knowledge. Proposed hydroelectric projects were obtained from *Eletrobrás* (4).

3) *Fundação Instituto Brasileiro de Geographifia e Estatistica (IBGE), 1995. Amazônia Legal. 1:3,000,000* Parada de Lucas, Rio de Janeiro.

4) *Eletrobrás: The Ten-Year Expansion Plan, B, 1998-2007 (Centrais Elétricas do Brasil, Rio de Janeiro, Brazil, 1998).*

Total Present Hydroelectric Area;	699,712 ha
Total 2011 Planned Hydroelectric Area;	7,832,409 ha
Total 2021 Planned Hydroelectric Area;	1,421,822 ha

Use in Models: Forested Areas within 10 km of the open surface water of a hydroelectric project are classified as lightly degraded. Forested Areas within 5 km are moderately degraded. Forested areas in the surface waters are classified as heavily degraded.

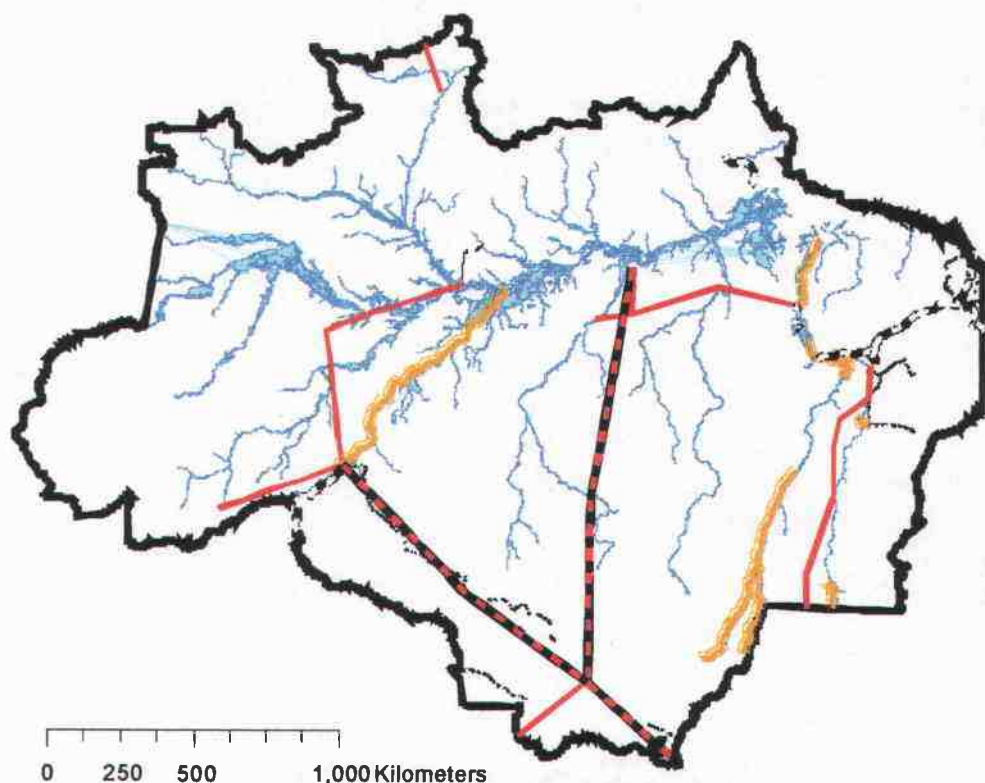


Figure C.5 Present and proposed infrastructure improvements within the Brazilian Legal Amazon with present utility corridors (black line), proposed utility corridors (red line), present railroads (white and black hatch line), proposed railroads (red and black hatch line) and proposed river channelization projects (orange triple line).

Source: Existing infrastructure projects were from 1995 IBGE map of Brazilian Legal Amazon (3) and personal knowledge. Planned infrastructure projects were from Maps and information provided by *Avança Brasil* (1), *Brasil em Ação* (2), *Eletrobrás* (4), and personal knowledge.

Total Present Utility Corridors: 3,388 km

Proposed Utility Corridors: 3,701 km

Total Present Railroad: 1,566 km

Proposed Railroads : 2,954 km

Proposed River Channelization Projects: 2,864 km

Use in the Model: Areas within 50 km of utility corridors, railroads or river channelization projects are classified as lightly degraded. Areas within 25 km of these projects are moderately degraded. Areas that are within 10 km of projects are heavily degraded.

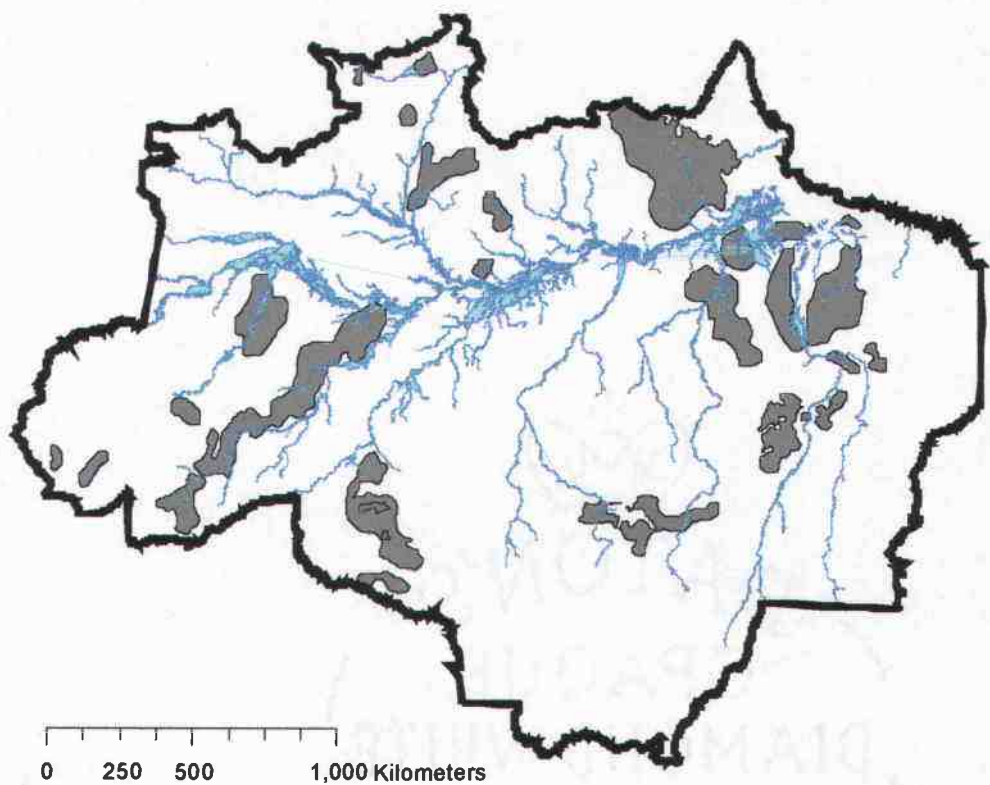


Figure C.6 Timber concessions occurring within the Brazilian Legal Amazon as of the year 2001.

Source: 1998 map of estimated legal and illegal logging, produced by IBAMA(5), Brazil’s national environmental agency

Total Area Under Harvest: 52,527,216 ha

Use in Models: Areas that are within timber concessions are classified as moderately degraded.

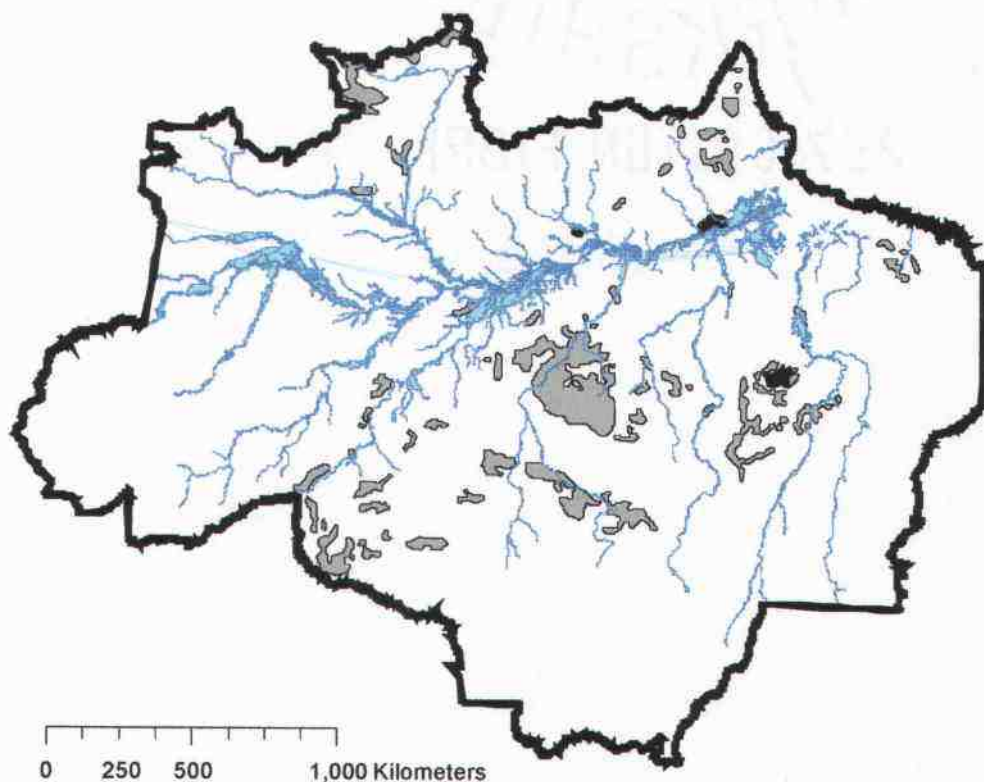


Figure C.7 Spatial distribution of federally approved (black) and 'wildcat' mining operations (grey) occurring within the Brazilian Legal Amazon (2001).

Source : 1998 map of estimated wild-cat gold mining and industrial mining, produced by IBAMA(5), Brazil's national environmental agency.

Total Area of Industrial Mining: 1,059,339 ha

Total Area of Wildcat Mining: 22,844,496 ha

Use in Model: Areas that occur within 'wildcat' or recognized mining facilities are classified as lightly degraded.

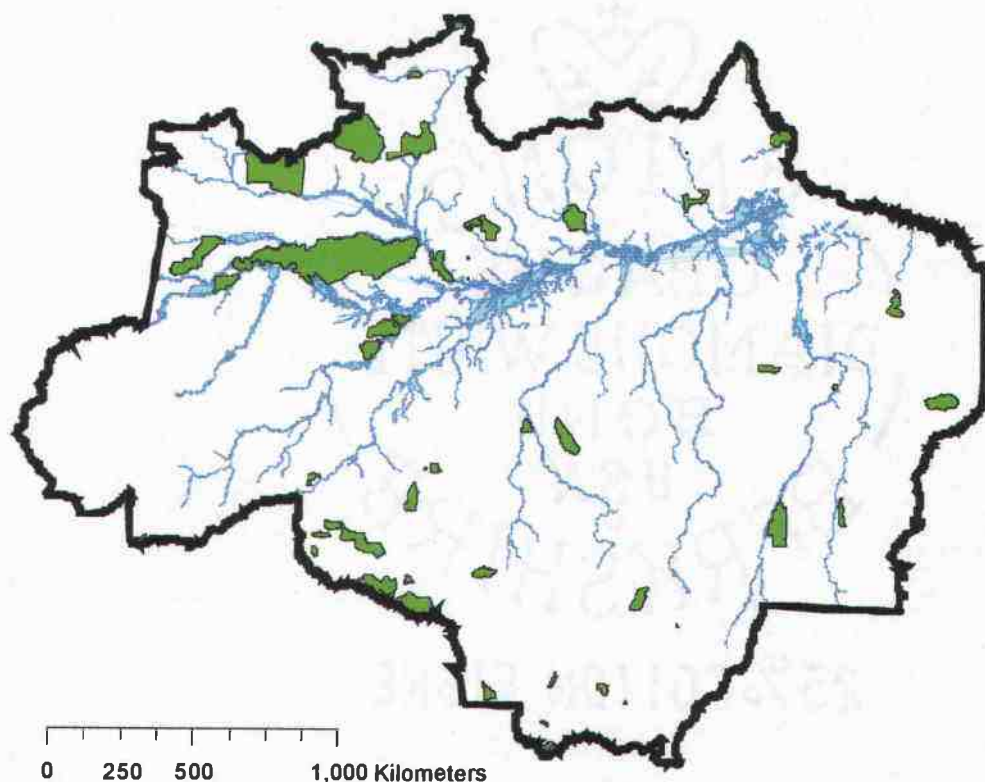


Figure C.8 Spatial extent of Ecological Reserves with a high degree of protection that occur within the Legal Brazilian Amazon.

Source: Ecological high protection reserves are the combined reserves of National and state parks, Ecological reserves, biological reserves and ecological stations within the Legal Brazilian Amazon. Table C.1 lists activities permitted and banned under specific reserve types. High protection reserves were delineated from 1995 IBGE map of Brazilian Legal Amazon, supplemented by 1999 map of Amazonian protected areas and personal knowledge. Some reserves protect aquatic regions. Recent private reserves are not included within this delineation.

Total Area of High Protection Reserves: 22,047,929 ha

Use in Model: Forested high protection reserves are classified as pristine even if they occur within buffers or other degraded habitat areas.

Table C.1 Activities permitted within high protection reserves.

Type of Area	Recreation& Tourism	Agriculture & Livestock	Logging	Harvesting	Hunting	Mining
National/state parks	Yes	No	No	No	No	No
Ecological reserves	Yes	No	No	No	No	No
Biological reserves	No	No	No	No	No	No
Ecological stations	No	No	No	No	No	No

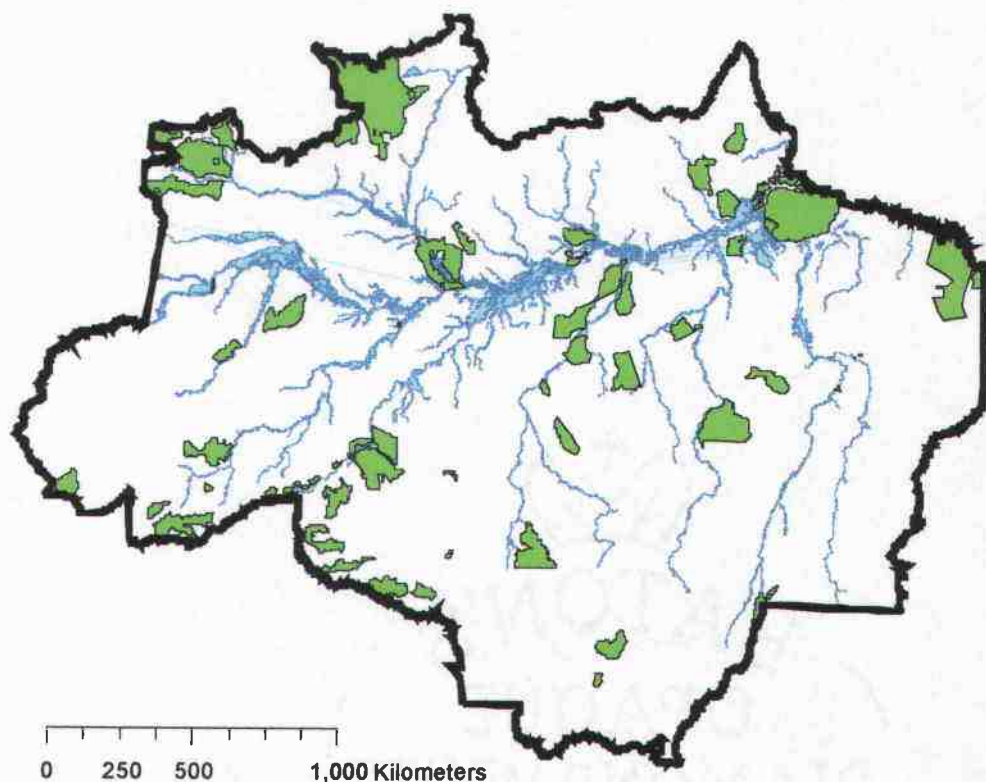


Figure C.9 Spatial distribution of moderately protected state administered sustainability reserves occurring within the Legal Brazilian Amazon.

Source: State administered reserves (moderate protection) are the combination of National/state forests, National forest reserves, Extractive reserves, State extractive forests, Sustainable-use forests, Sustainable development Reserves, Environmental protection areas and Areas of relevant ecological interest. Table C.2 lists activities permitted and banned under specific reserve systems. Moderate protection reserves were delineated from 1995 IBGE map of Brazilian Legal Amazon, supplemented by 1999 map of Amazonian protected areas and personal knowledge. Some reserves protect aquatic regions. Recent private reserves are not included within this delineation.

Total Area with Moderate Protection: 40,795,308 ha

Use in Model: Moderately protected forested areas were classified as lightly degraded. For areas that exist within moderate protection reserves that were classified as heavily or moderately degraded, these areas were reclassified as lightly degraded.

Table C.2 Activities permitted within moderate protection reserves.

Type of Area	Recreation & Tourism	Agriculture & Livestock	Logging	Harvesting	Hunting	Mining
National/state forests	Yes	Yes	Yes	Yes	Yes ¹	No
National forest reserves	Yes	Yes	Yes	Yes	Yes ¹	No
Extractive reserves	Yes	Yes	Yes	Yes	Yes ¹	No
State extractive forests	Yes	Yes	Yes	Yes	Yes ¹	No
Sustainable-use forests	Yes	Yes	Yes	Yes	Yes ¹	No
Sustainable development Reserves	Yes	Yes	Yes	Yes	Yes ¹	No
Environmental protection areas	Yes	Yes ²	Yes ²	Yes ²	No	Yes ²
Areas of relevant ecological interest	Yes	Yes ²	No	Yes ²	No	No

¹Hunting is allowed in some areas; for others information was unavailable.

²These activities are not expressly permitted, but because people are allowed to live in these reserves they will certainly occur, at least on a limited scale.

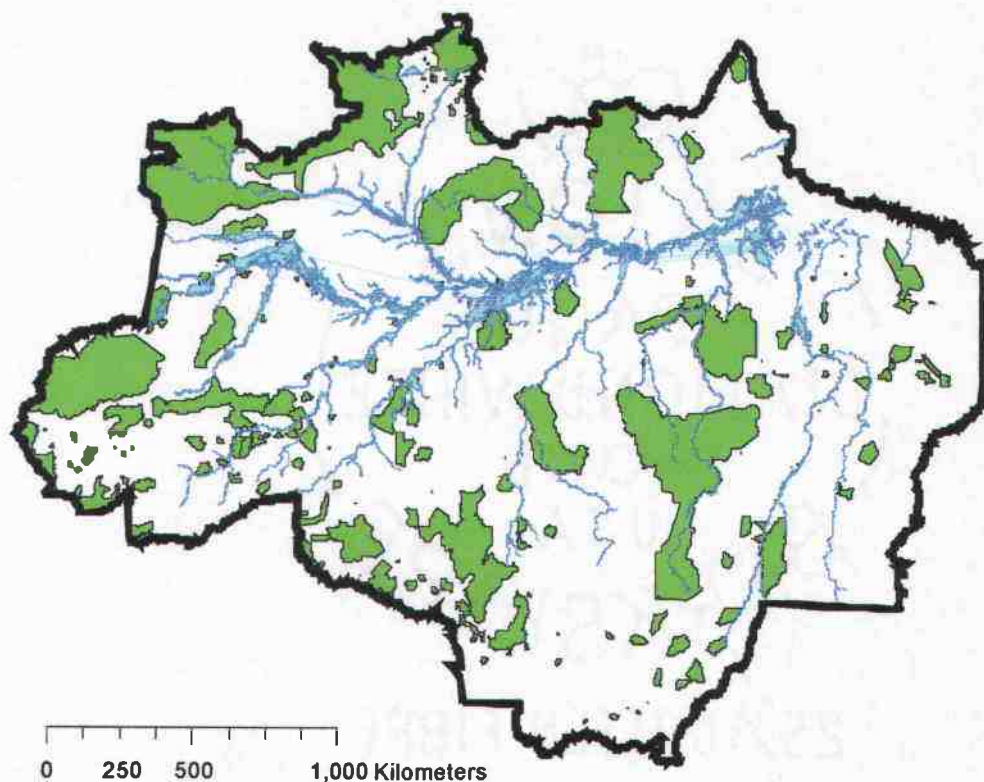


Figure C.10 Spatial distribution of delineated indigenous people reserves occurring within the Brazilian Legal Amazon (2001).

Source: Indigenous lands were delineated from 1995 IBGE map of Brazilian Legal Amazon, supplemented by 1999 map of Amazonian protected areas and personal knowledge. Indigenous lands exclude recreation, tourism and mining but allow for agriculture, logging, harvesting and hunting, presumably for subsistence purposes.

Total Area within Indigenous Lands: 102,130,590 ha

Use in Models: Forested areas that occurred in areas that were previously classified as heavy or moderately degraded were reclassified as lightly degraded if they occurred within indigenous people's territories.

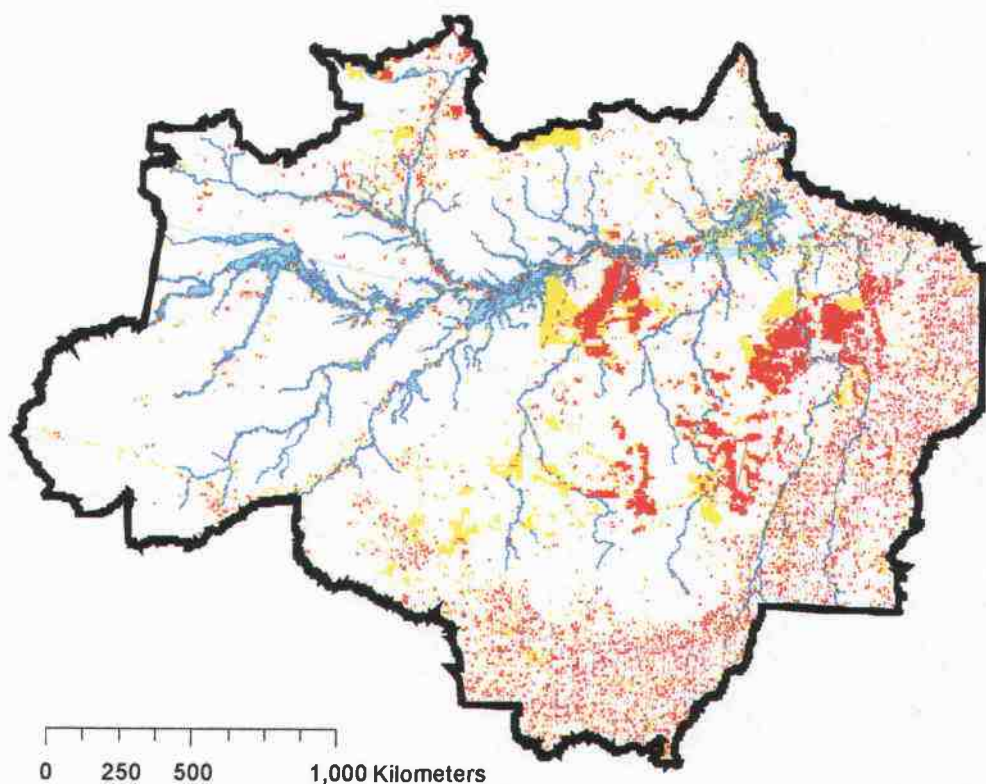


Figure C.11 Spatial extent of high (red) and moderate (yellow) fire potential existing within the Brazilian Legal Amazon (Nepstad *et al.*, 1998).

Source: Map of areas with high, medium, and low fire vulnerability, based on analyses of forest cover, seasonal soil moisture, logging activity, and recent fires during the 1998 dry season (6).

6) It has been estimated that about 200,000 km² of Brazilian Amazonian forest is vulnerable to fires during normal years, but this figure may approach 1.5 million km² during periodic El Niño droughts [D. C. Nepstad *et al.*, *Conserv. Biol.* 12, 951 (1998)].

Total Area with High Fire Potential: 25,723,847 ha

Total Area with Moderate Fire Potential: 16,171,852 ha

Use in Model: Forested areas that have a high fire potential are classified as moderately degraded. Forested areas with a moderate fire potential are classified as lightly degraded.

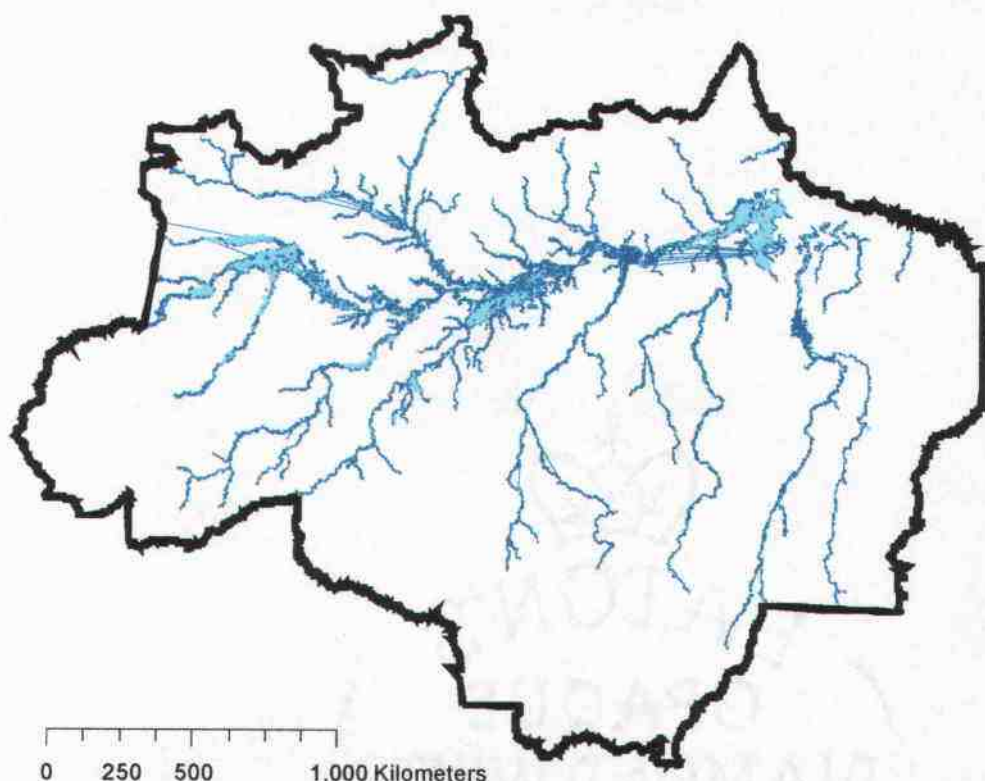


Figure C.12 Rivers and aquatic systems in which susceptibility to riverside deforestation occurring within the Brazilian Legal Amazon. Open water section (dark blue) is distinguished from the seasonally inundated areas (light blue).

Source: River coverage produced by the National Oceanographic and Atmospheric Administration based on 1999 AVHRR imagery in conjunction with IBGE digital maps for individual states within the Legal Brazilian Amazon. Aquatic systems that were wider than 1 km were determined to be navigable. Seasonally inundated forests were delineated by IBGE. The boundary of the Brazilian Legal Amazon was delineated by IBGE, 1995.

Total Area of Open Water: 8,316,532 ha

Total Area of Inundated Forests: 8,607,460 ha

Total Area of the Brazilian Legal Amazon: 503,270,922 ha

Use in Model: The model is only for the Brazilian Legal Amazon, all areas beyond this boundary were excluded from study. Forested areas within 10 km of navigable water are classified as lightly degraded. Areas within 5 km are moderately degraded. Areas within 2 km of navigable waters are classified as highly degraded.

APPENDIX D RESULTS OF THE SPATIAL MODELS USING ELEMENT ADDED AND ELEMENT DELETED
METHODOLOGIES

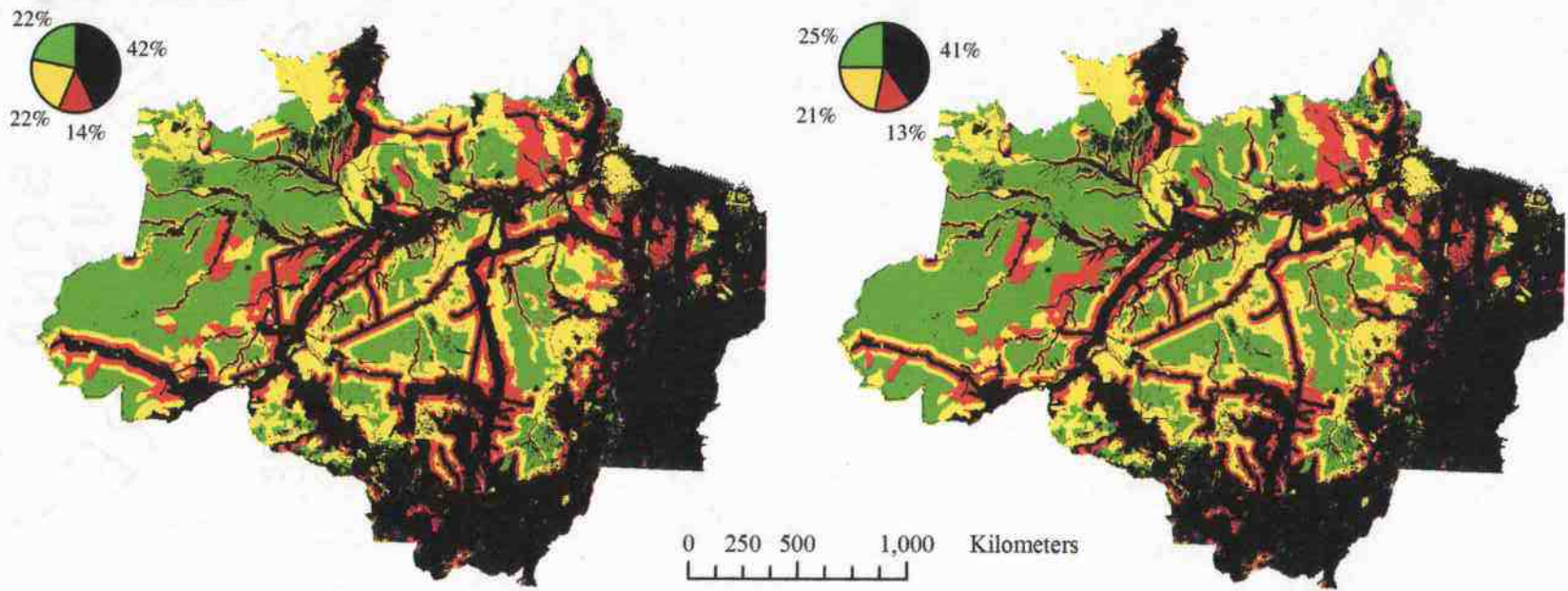


Figure D.1 Full and Null models depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.1 Area of each class type within the Full and Null model scenarios. Statistics are given in km².

	Full	Null
Class 1	2,157,499	2,017,619
Class 2	680,266	673,379
Class 3	1,098,189	1,081,253
Class 4	1,096,752	1,260,454

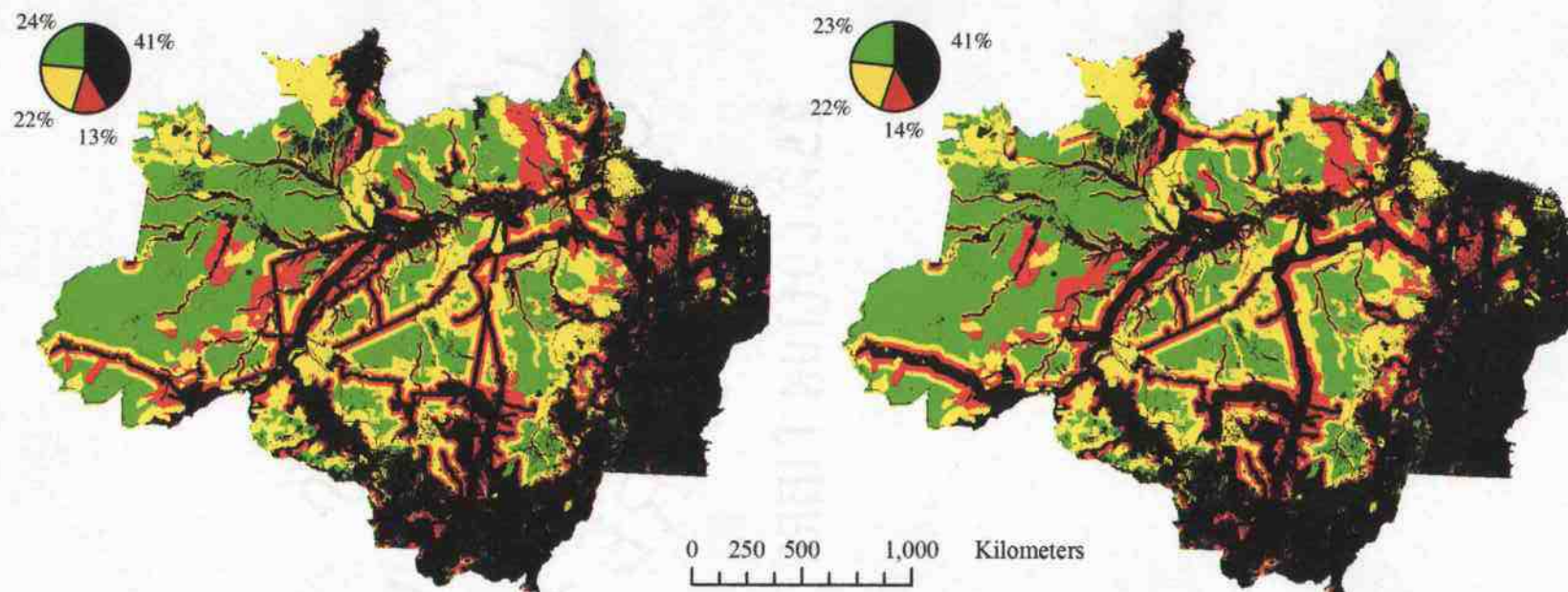


Figure D.2 Full model minus future road developments and the Null model plus all future road developments depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.2 Area of each class type within the ‘Full model minus all future road developments’ and the ‘Null model plus all future road developments’ scenarios. Statistics are given in km².

	Full – Road developments	Null + Road developments
Class 1	2,077,086	2,102,061
Class 2	678,598	679,604
Class 3	1,084,944	1,095,055
Class 4	1,192,078	1,155,986

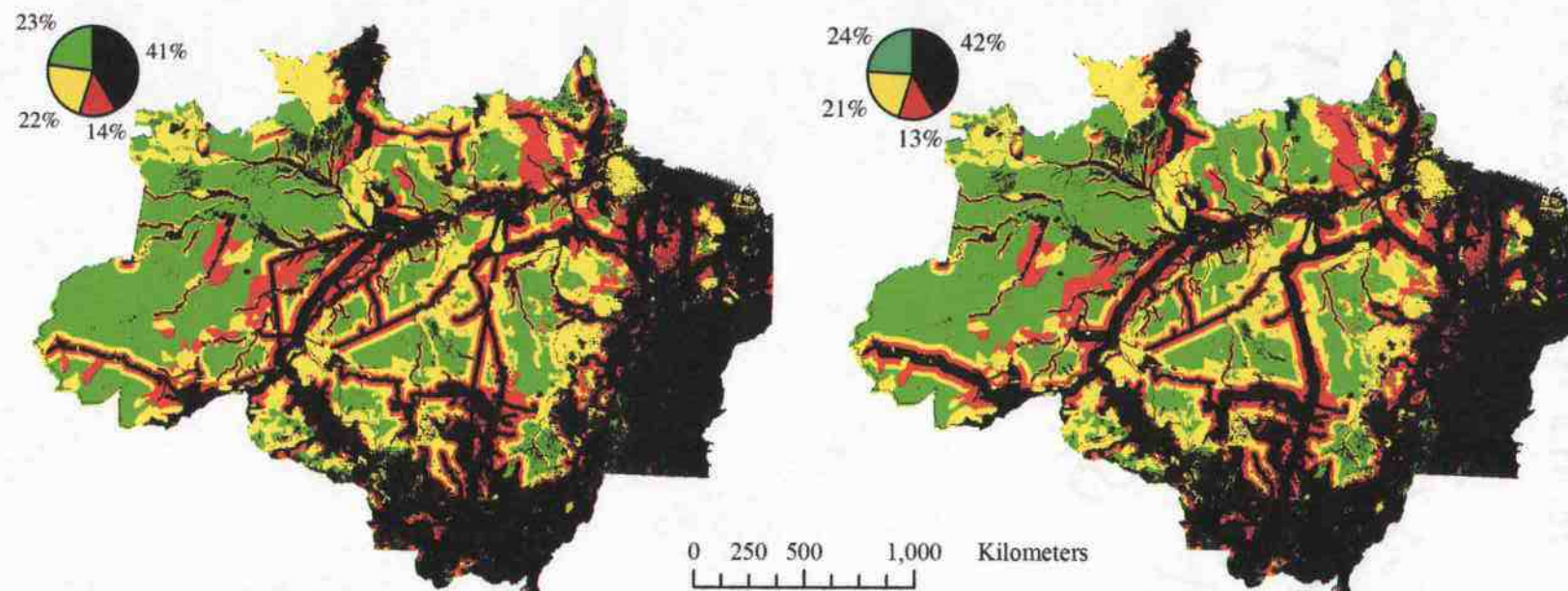


Figure D.3 Full model minus future road pavement upgrades and the Null model plus all future road upgrades depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.3 Area of each class type within the ‘Full model minus future road pavement upgrades’ and the ‘Null model plus all future road upgrades’ scenarios. Statistics are given in km².

	Full – Road Upgrades	Null + Road Upgrades
Class 1	2,087,914	2,091,448
Class 2	686,457	671,760
Class 3	1,114,007	1,065,800
Class 4	1,144,329	1,203,698

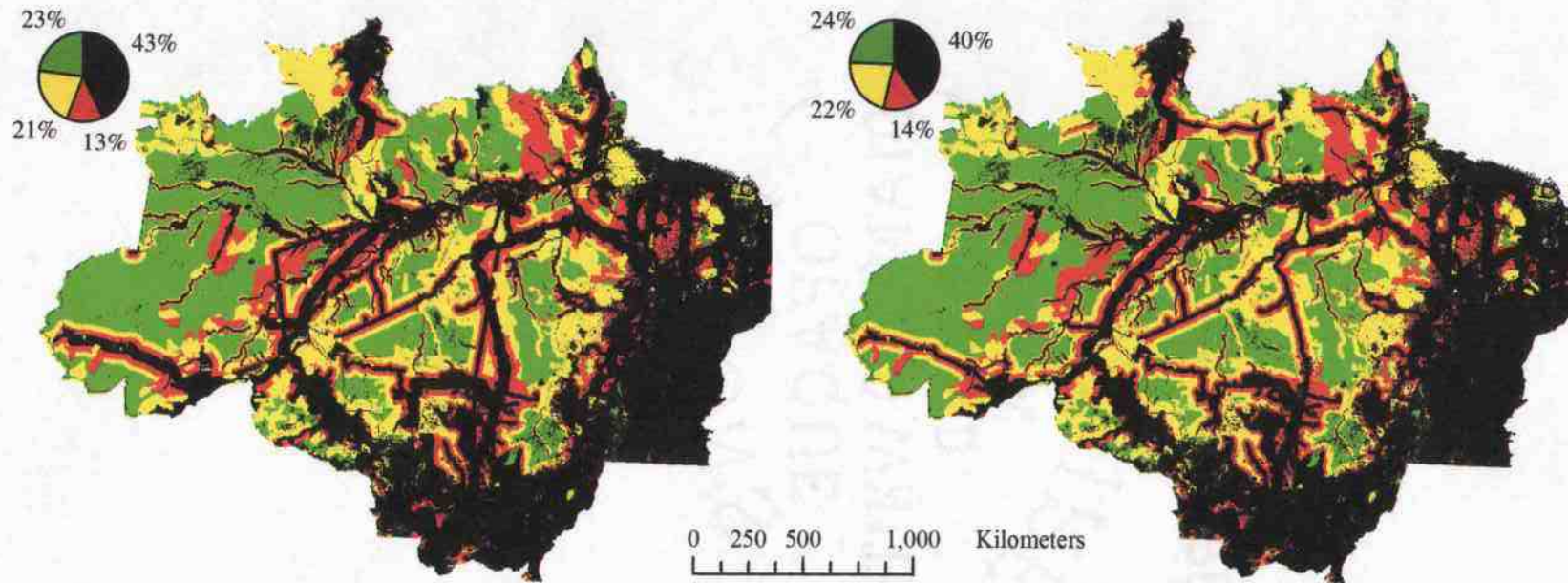


Figure D.4 Full model minus new roads and the Null model plus new roads depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.4 Area of each class type within the 'Full model minus new roads' and the 'Null model plus new roads' scenarios. Statistics are given in km².

	Full – New Roads	Null + New Roads
Class 1	2,087,914	2,028,447
Class 2	686,457	681,238
Class 3	1,114,007	1,110,315
Class 4	1,144,329	1,212,705

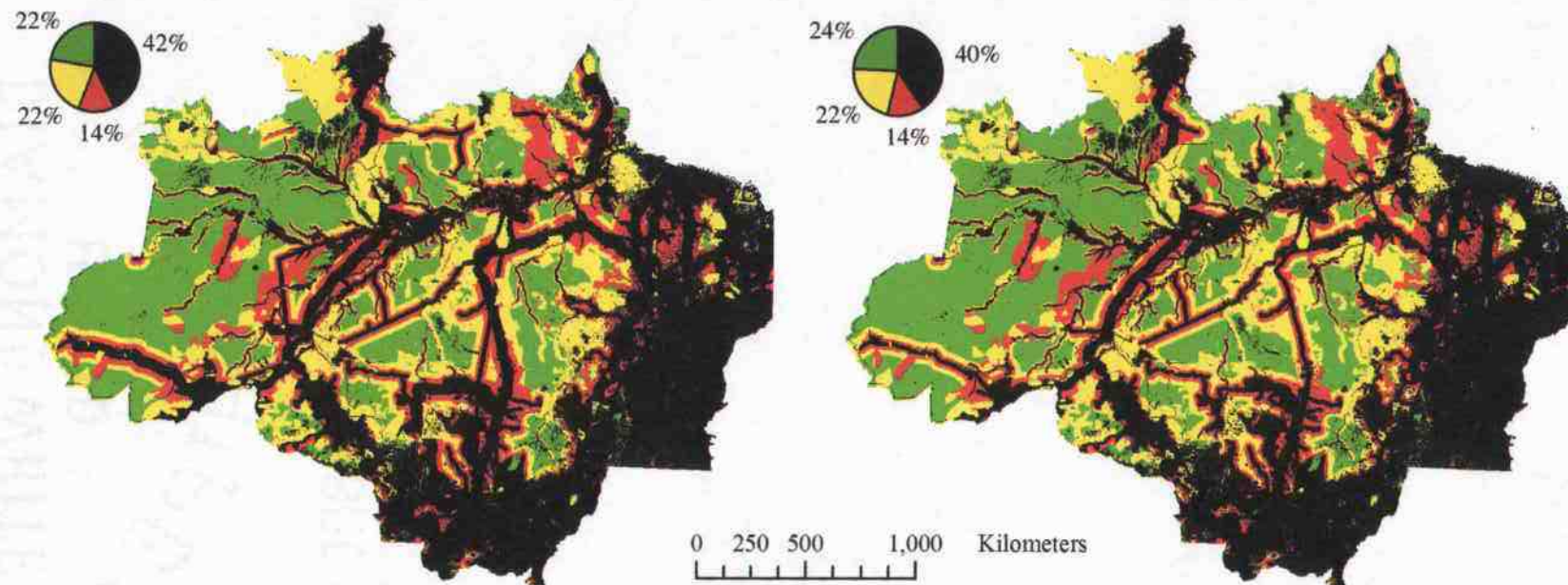


Figure D.5 Full model minus all new dams and the Null model plus all new dams depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.5 Area of each class type within the 'Full model minus all new dams' and the 'Null model plus all new dams' scenarios. Statistics are given in km².

	Full – All New Dams	Null + All New Dams
Class 1	2,138,692	2,033,039
Class 2	684,214	669,010
Class 3	1,093,369	1,085,493
Class 4	1,116,432	1,245,164

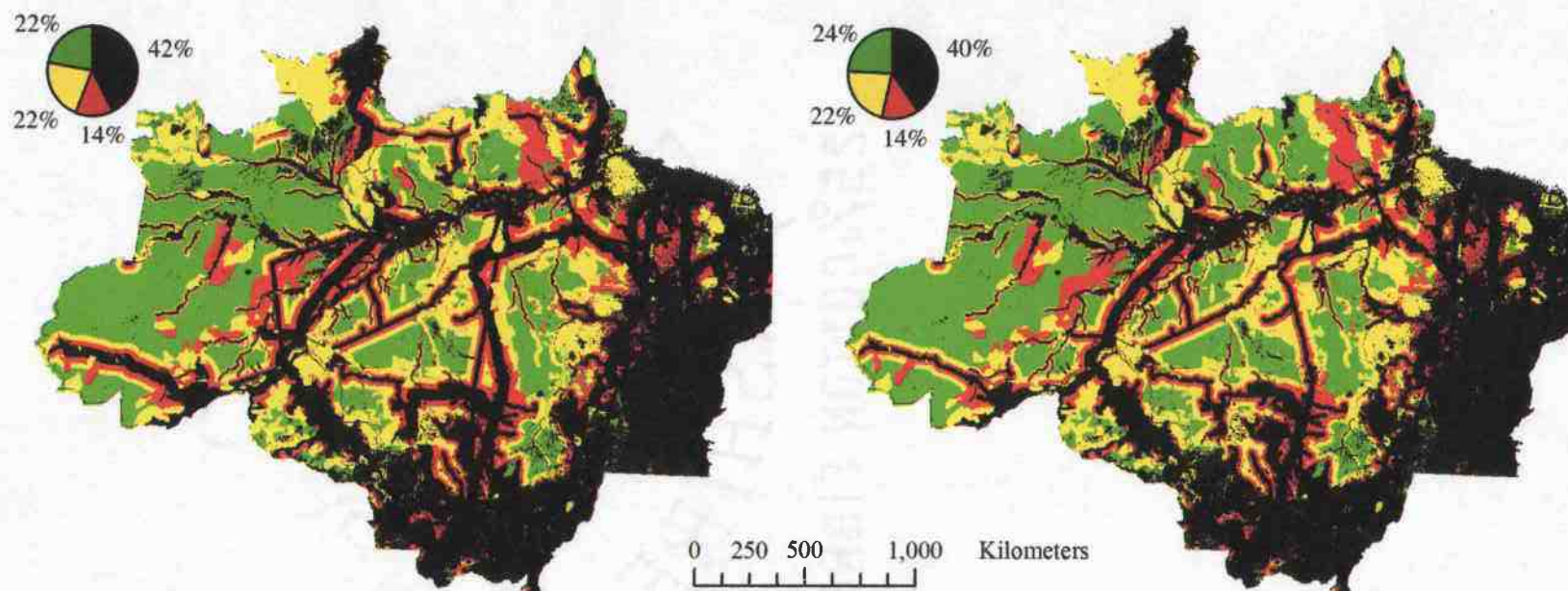


Figure D.6 Full model minus near future dams and the Null model plus near future dams depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.6 Area of each class type within the ‘Full model minus all early dams’ and the Null model plus all early dams’ scenarios. Statistics are given in Km².

	Full – Near Future Dams	Null + Near Future Dams
Class 1	2,142,591	2,029,140
Class 2	684,703	668,520
Class 3	1,095,907	1,082,955
Class 4	1,109,505	1,252,090

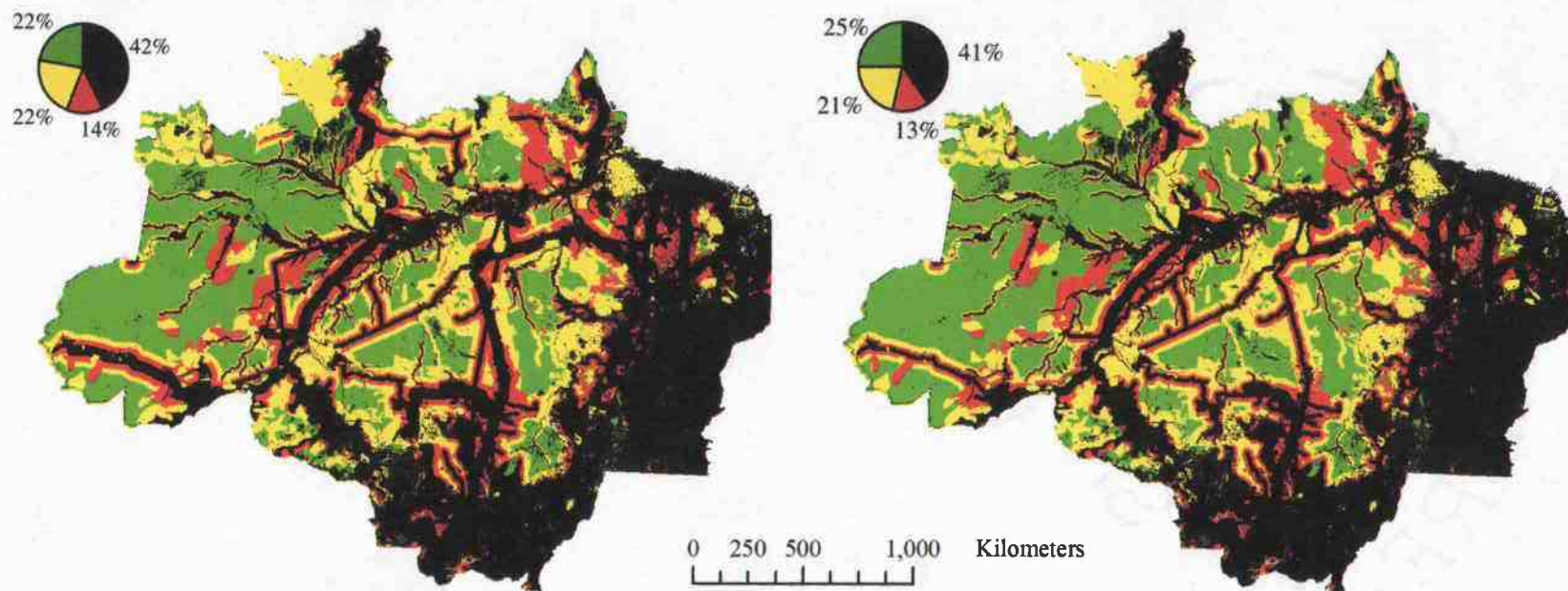


Figure D.7 Full model minus river channelization and the Null model plus river channelization depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.7 Area of each class type within the 'Full model minus river channelization' and the 'Null model plus river channelization' scenarios. Statistics are given in km².

	Full – Channelization	Null + Channelization
Class 1	2,147,774	2,027,504
Class 2	679,920	674,181
Class 3	1,101,974	1,076,854
Class 4	1,103,039	1,254,167

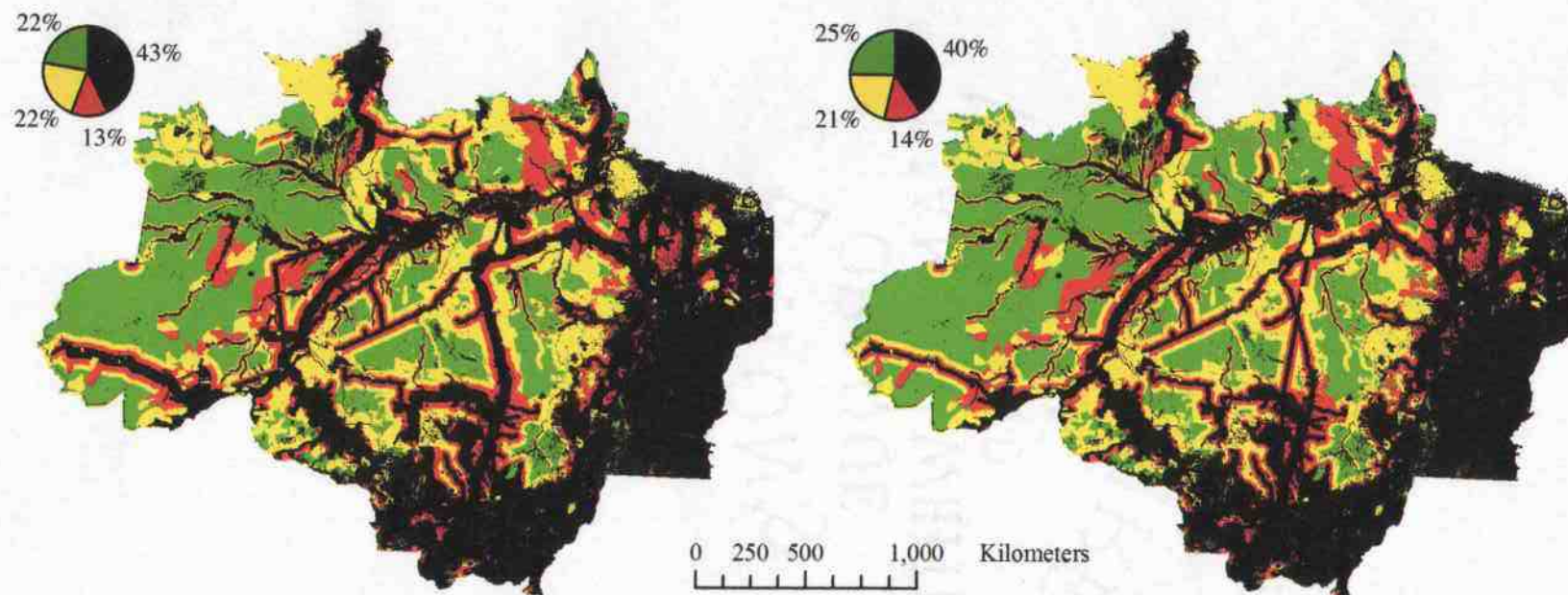


Figure D.8 Full model minus new railroads and the Null model plus new railroads depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.8 Area of each class type within the 'Full model minus new railroads' and the 'Null model plus new railroads' scenarios. Statistics are given in km².

	Full – New Railroads	Null + New Railroads
Class 1	2,146,006	2,030,799
Class 2	677,058	680,749
Class 3	1,101,383	1,078,919
Class 4	1,108,260	1,242,239

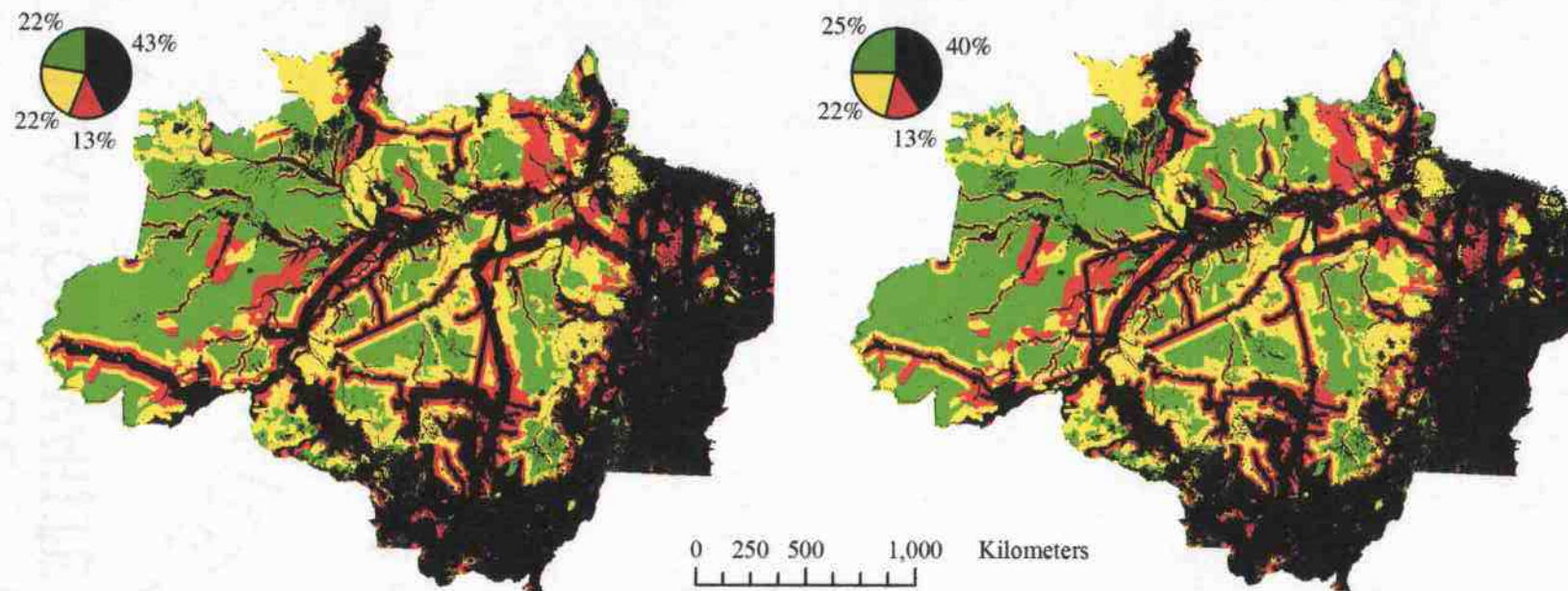


Figure D.9 Full model minus new utilities and the Null model plus new utilities depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.9 Area of each class type within the ‘Full model minus new utilities’ and the ‘Null model plus new utilities’ scenarios. Statistics are given in km².

	Full – New Utilities	Null + New Utilities
Class 1	2,142,087	2,034,881
Class 2	679,212	674,133
Class 3	1,093,521	1,085,913
Class 4	1,117,886	1,237,778

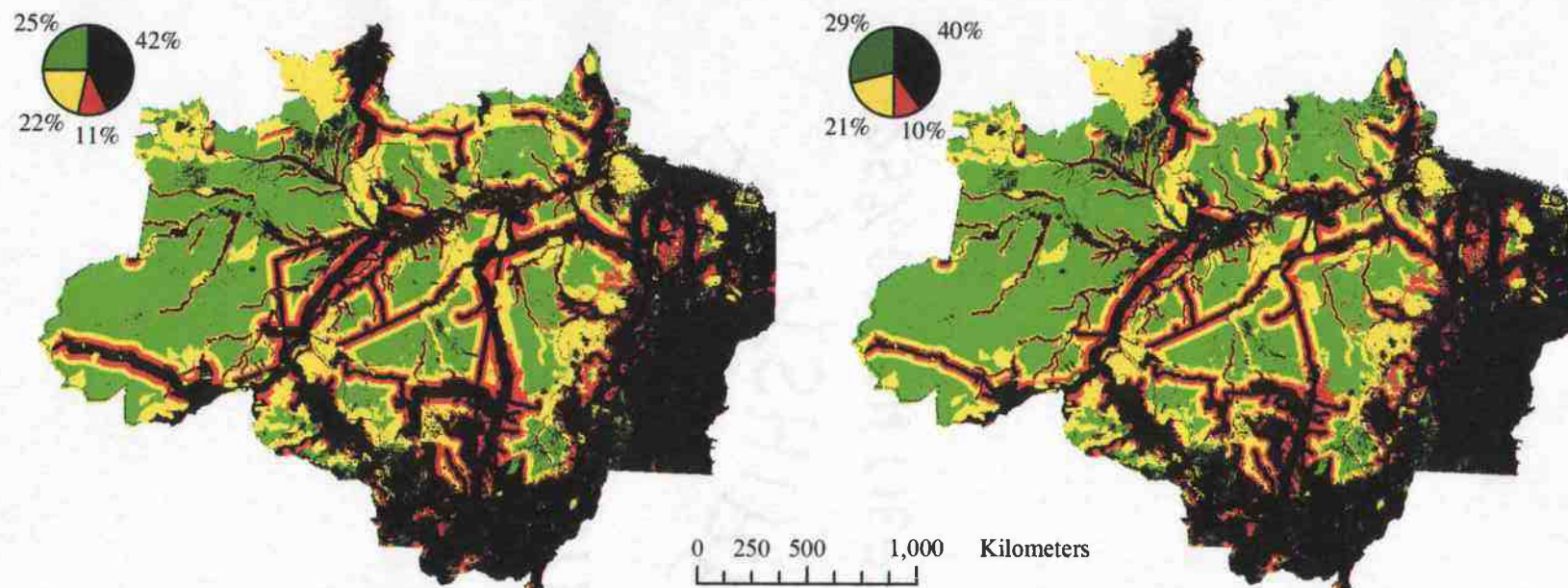


Figure D.10 Full model minus areas of natural exploitation and the Null model minus areas of natural exploitation depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.10 Area of each class type within the 'Full model minus areas of natural exploitation' and the 'Null model minus areas of natural exploitation' scenarios. Statistics are given in km².

	Full – Exploitation	Null - Exploitation
Class 1	2,157,499	2,017,619
Class 2	538,446	508,984
Class 3	1,085,295	1,045,419
Class 4	1,251,465	1,460,682

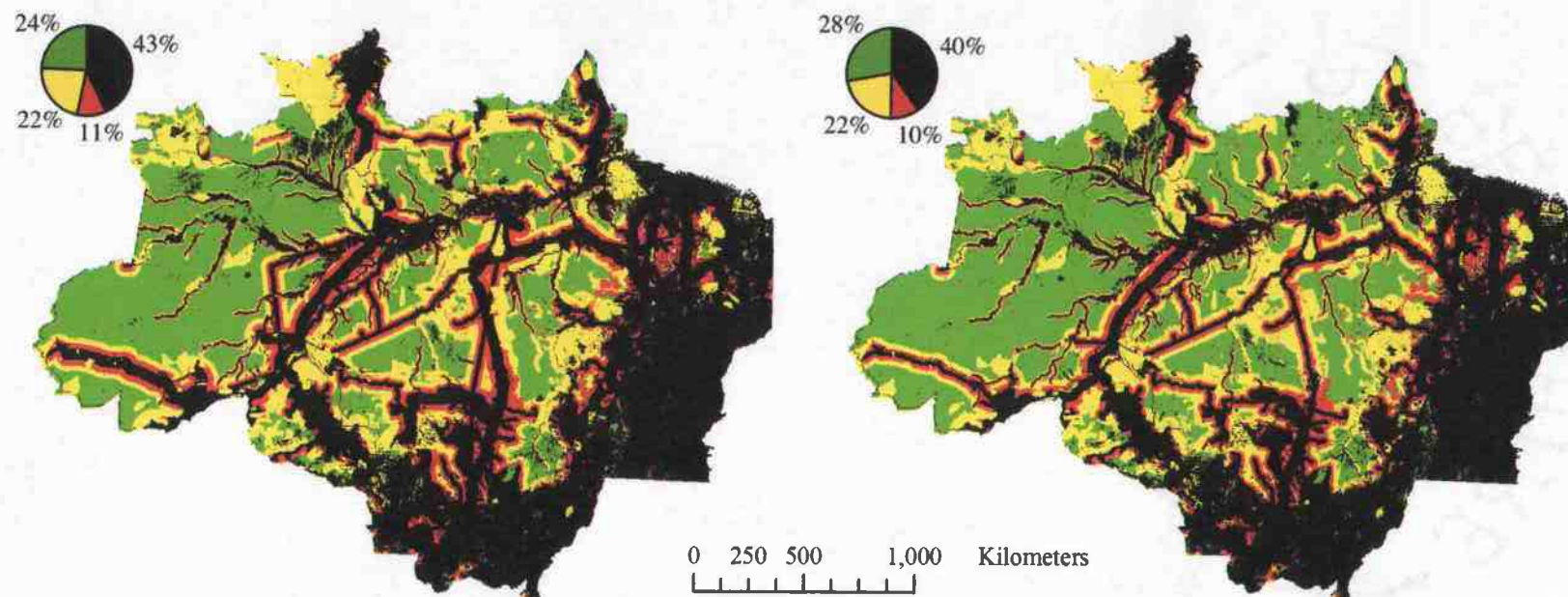


Figure D.11 Full model minus areas of timber concessions and the Null model minus areas of timber concessions depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.11 Area of each class type within the 'Full model minus areas of timber concessions and the Null model minus areas of timber concessions' scenarios. Statistics are given in km².

	Full – Timber Harvest	Null – Timber Harvest
Class 1	2,147,774	2,017,619
Class 2	538,100	508,984
Class 3	1,125,055	1,087,400
Class 4	1,221,777	1,418,701

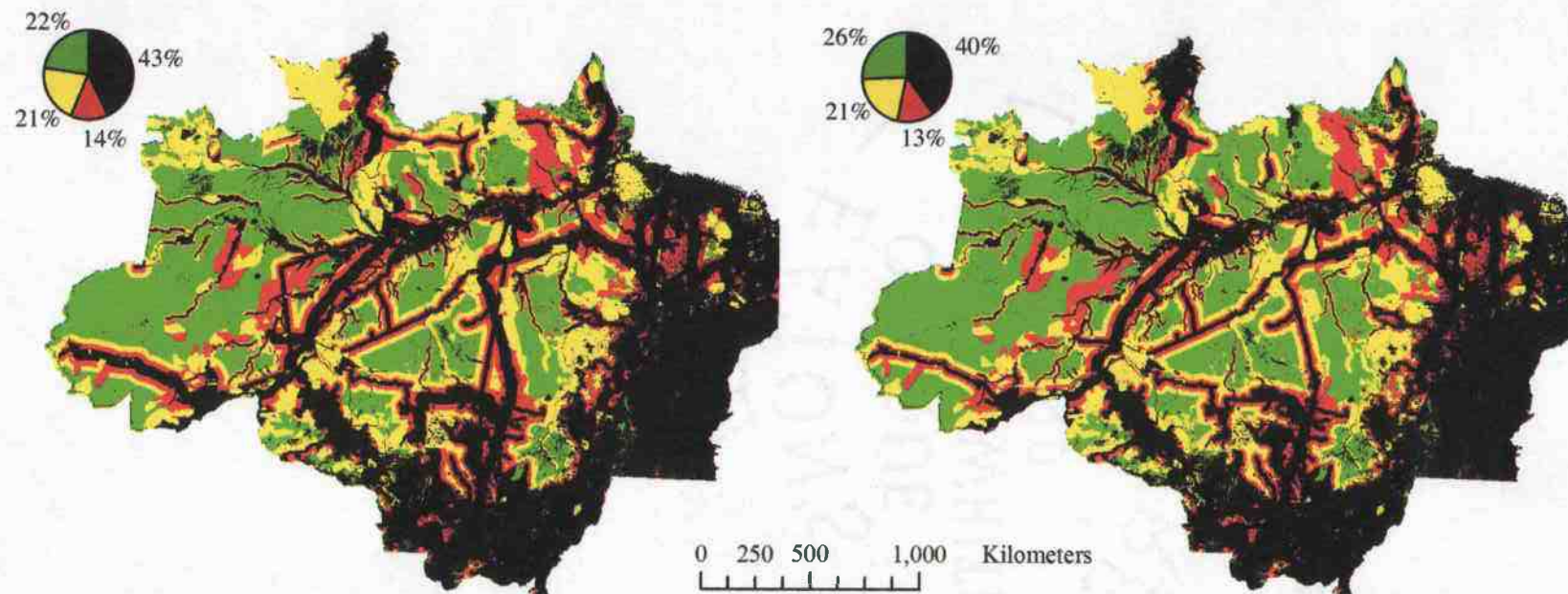


Figure D.12 Full model minus areas of mining and the Null model minus areas of mining depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.12 Area of each class type within the ‘Full model minus areas of mining’ and the ‘Null model minus areas of mining’ scenarios. Statistics are given in km².

	Full – Mining	Null – Mining
Class 1	2,157,499	2,017,619
Class 2	680,266	673,379
Class 3	1,063,340	1,040,397
Class 4	1,131,601	1,301,309

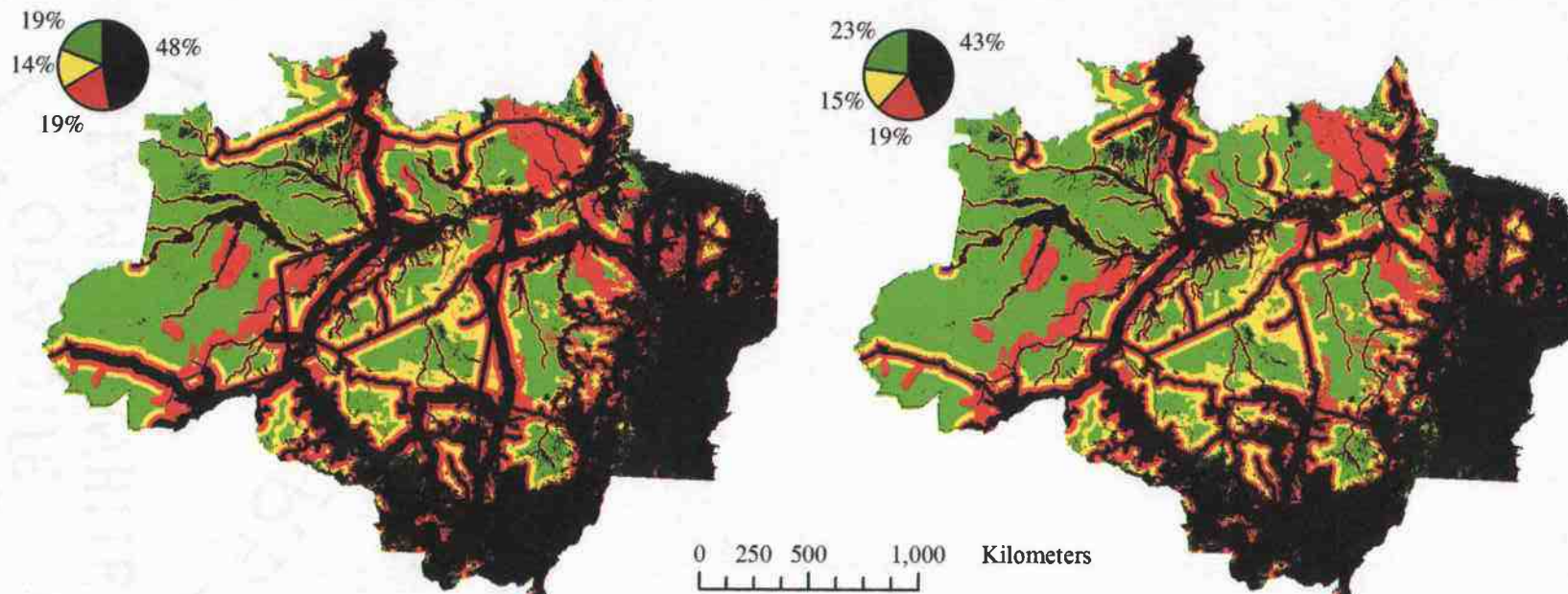


Figure D.13 Full model minus areas of reserve systems and the Null model minus areas of reserve systems depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.13 Area of each class type within the 'Full model minus areas of reserve systems' and the 'Null model minus areas of reserve systems' scenarios. Statistics are given in km².

	Full – Reserve Systems	Null – Reserve Systems
Class 1	2,361,071	2,169,396
Class 2	975,950	950,649
Class 3	726,624	767,376
Class 4	969,061	1,145,283

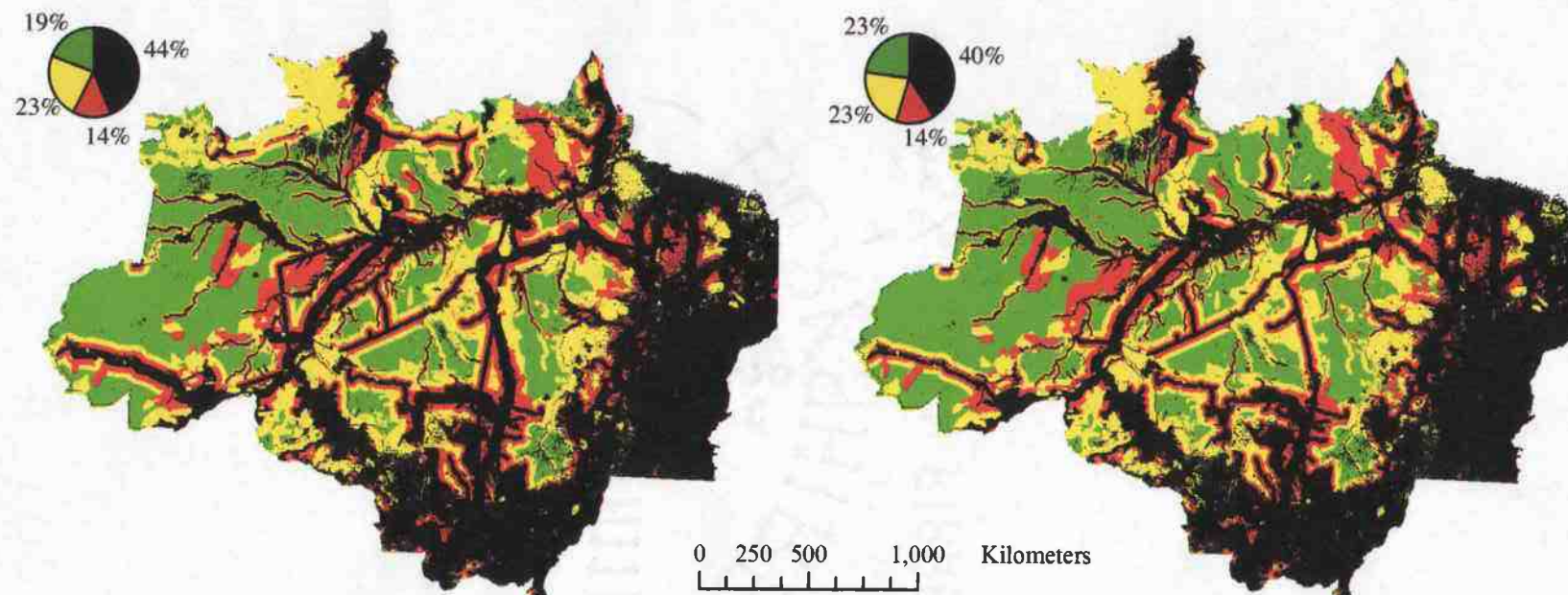


Figure D.14 Full model minus areas of high reserve protection and the Null model minus areas of high reserve protection depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.14 Area of each class type within the 'Full model minus areas of high reserve protection' and the 'Null model minus areas of high reserve protection' scenarios. Statistics are given in km².

	Full – High Protection	Null – High Protection
Class 1	2,191,235	2,049,258
Class 2	709,444	700,057
Class 3	1,162,966	1,138,107
Class 4	969,061	1,145,283

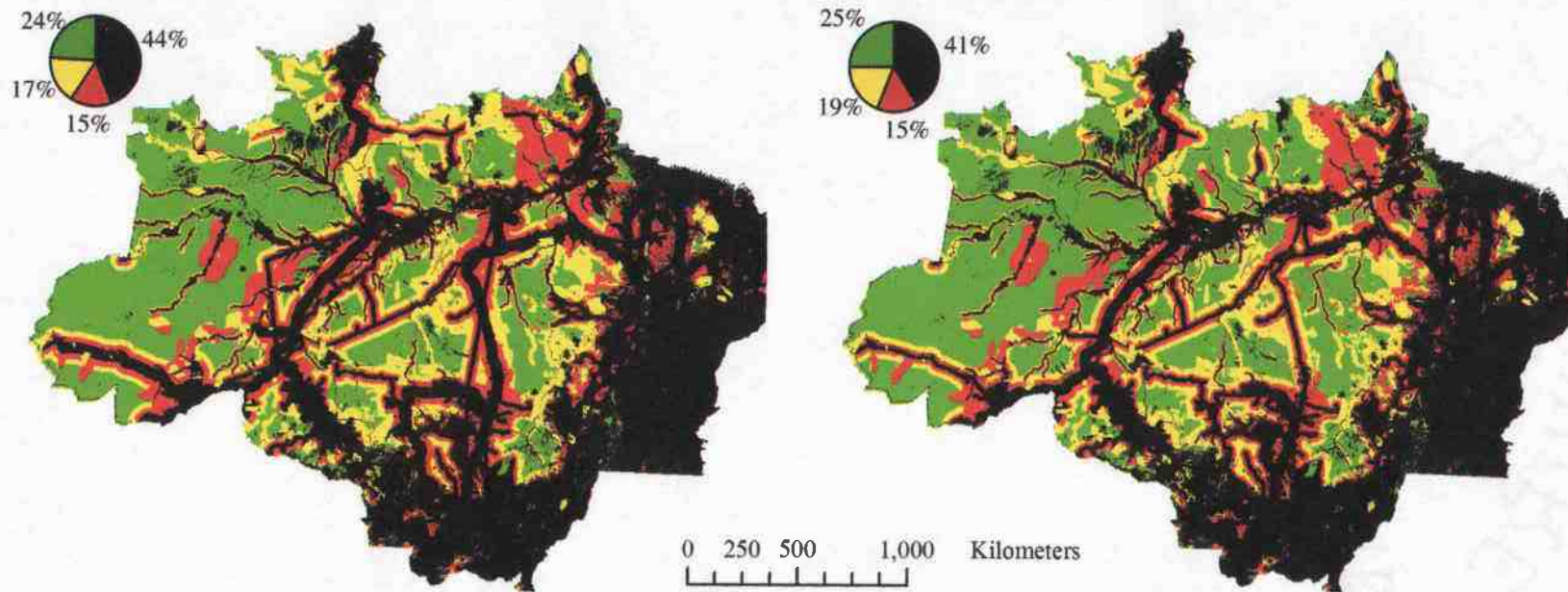


Figure D.15 Full model minus areas of moderate reserve protection and the Null model minus areas of moderate reserve protection depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.15 Area of each class type within the 'Full model minus areas of moderate reserve protection' and the 'Null model minus areas of moderate reserve protection' scenarios. Statistics are given in km².

	Full – Mod. Protection	Null – Mod. Protection
Class 1	2,214,547	2,064,502
Class 2	761,901	754,999
Class 3	860,190	952,750
Class 4	1,196,068	1,260,454

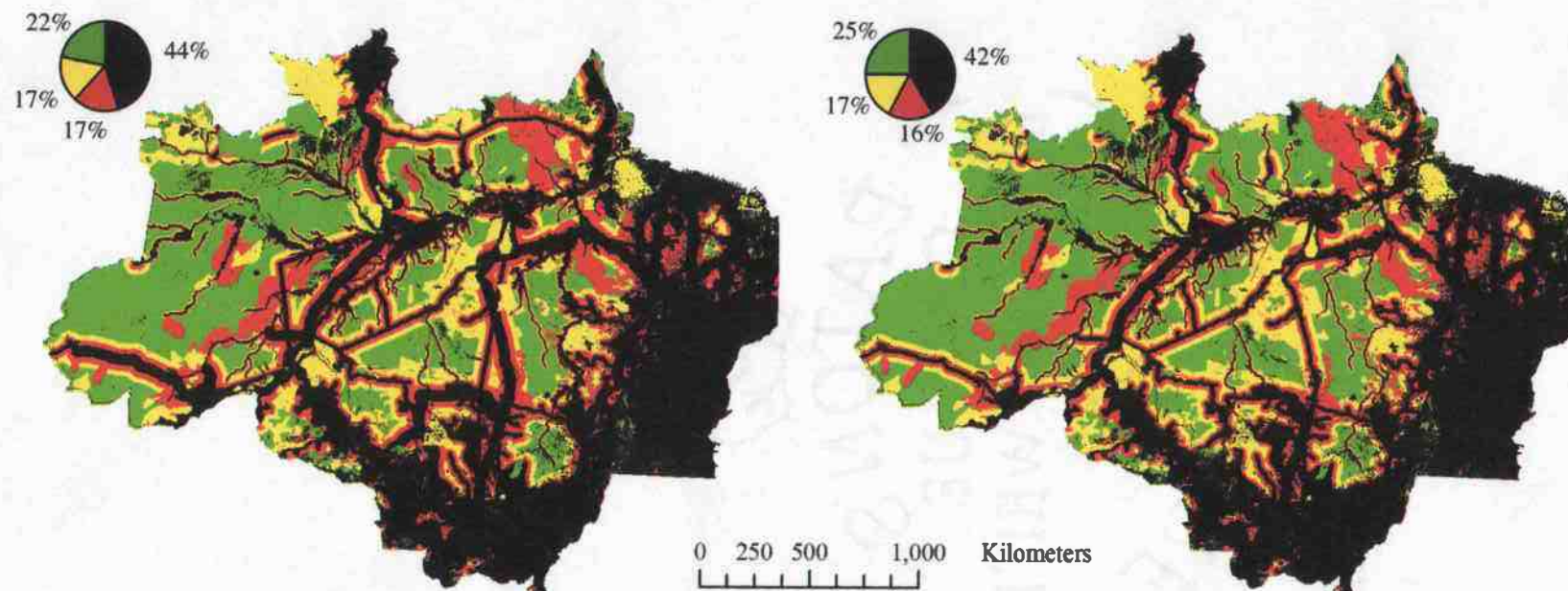


Figure D.16 Full model minus areas of indigenous nation protection and the Null model minus areas of indigenous nation protection depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.16 Area of each class type within the 'Full model minus areas of indigenous nation protection' and the 'Null model minus areas of indigenous nation protection' scenarios. Statistics are given in km².

	Full – Indigenous	Null – Indigenous
Class 1	2,258,980	2,084,096
Class 2	843,409	823,012
Class 3	833,565	865,143
Class 4	1,096,752	1,260,454

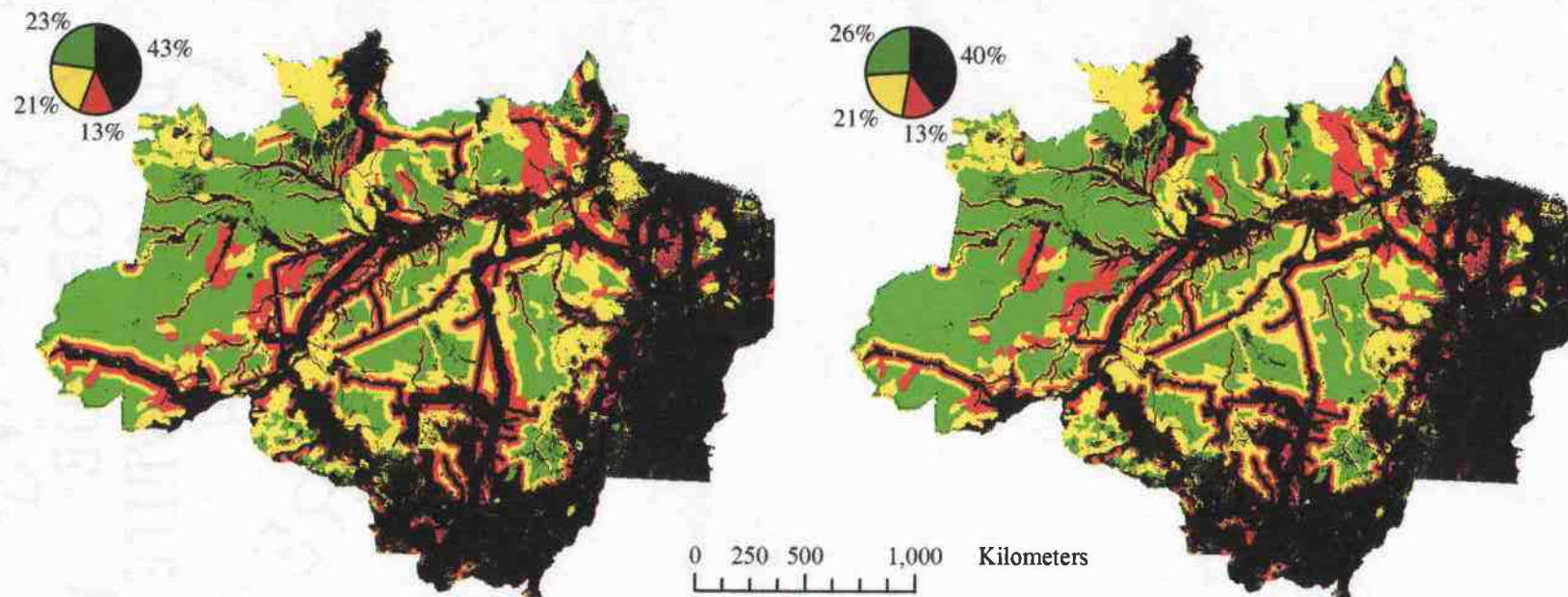


Figure D.17 Full model minus areas prone to fire and the Null model minus areas prone to fire depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.17 Area of each class type within the 'Full model minus areas prone to fire' and the 'Null model minus areas prone to fire' scenarios. Statistics are given in km².

	Full – Fire	Null – Fire
Class 1	2,157,499	2,017,619
Class 2	658,731	643,243
Class 3	1,074,050	1,051,433
Class 4	1,142,429	1,320,413

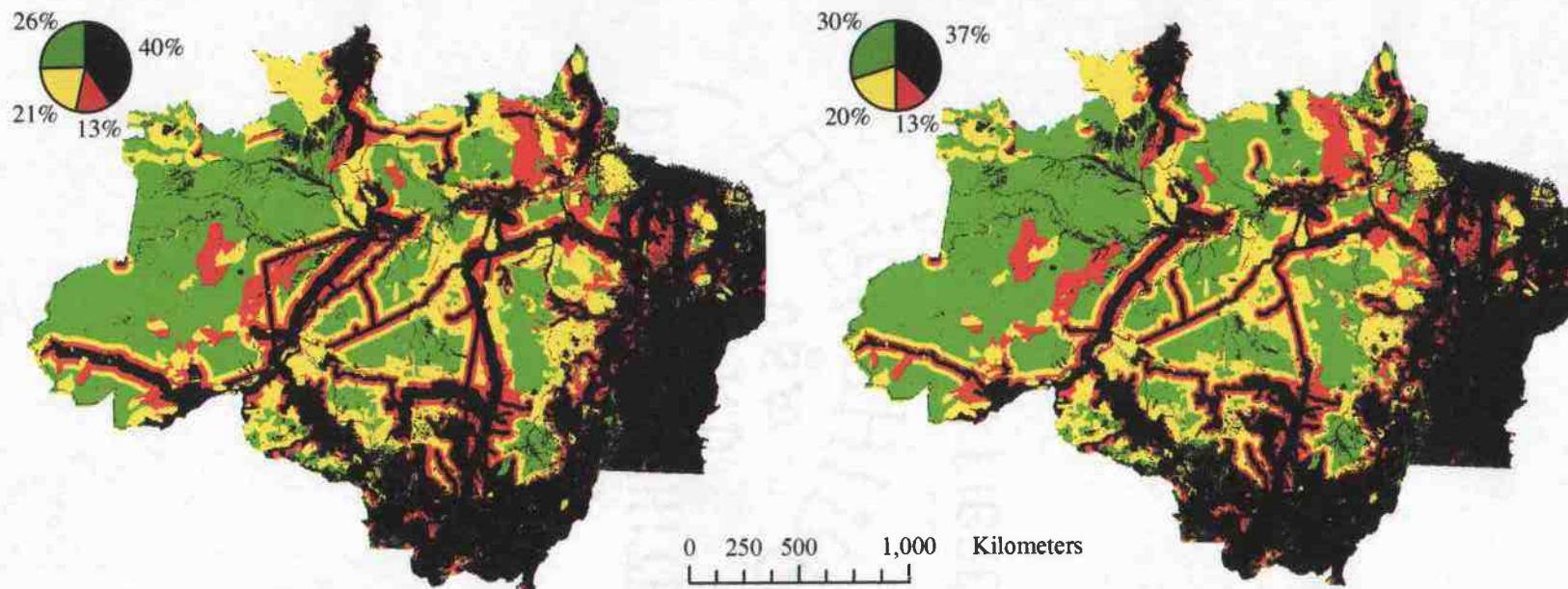


Figure D.18 Full model minus river deforestation and the Null model minus river deforestation depicting pristine (green), lightly degraded (yellow), moderately degraded (red) and heavily degraded forest habitats (black). Pie charts for the composition of the models are represented in pie charts with respective colors (upper left of model).

Table D.18 Area of each class type within the ‘Full model minus river deforestation’ and the ‘Null model minus river deforestation’ scenarios. Statistics are given in km².

	Full – River Deforest.	Null – River Deforest.
Class 1	2,031,816	1,874,562
Class 2	666,322	646,803
Class 3	1,041,653	1,000,509
Class 4	1,292,914	1,510,830