Land-use change in the Neotropics: regional-scale predictors of deforestation and local effects on carbon storage and tree-species diversity

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Preface

The two core chapters in this thesis represent two independent studies that are related by their common theme. The first has been submitted for publication, and the second is being prepared for submission. These core chapters are supplemented by an introduction, conclusion and appendices that provide a more complete context for the work. The manuscript format of my thesis is in accordance with McGill University's formal thesis requirements: "As an alternative to the traditional thesis format, the dissertation can consist of a collection of papers of which the student is an author or co-author." (available at www.mcgill.ca/gps/programs/thesis/guidelines/preparation).

Contribution of Authors

Chapter 2 expands on a study published in 2002 by Drs. William Laurance, Ana Albernaz, Götz Schroth, Philip Fearnside, Scott Bergen, Eduardo Venticinque and Mr. Carlos da Costa. The co-authors collected the data on deforestation and on the various predictor variables, and developed the GIS database upon which my analyses are based. My study is a sensitivity analysis of the results of Laurance et al. (2002). I worked with Dr. Laurance to develop the idea for the analysis, and I was responsible for designing the sampling methodology, carrying out the sampling, analyzing the data, interpreting the results and writing the manuscript. Discussions with Dr. Laurance and Dr. Albernaz contributed to each stage of the process, and feedback, suggestions and editing from all of the co-authors contributed to the final paper. (See Appendix 3 for signed waivers from co-authors permitting me to include the manuscript in my thesis).

Chapter 3 was written in cooperation with Dr. Catherine Potvin. I was responsible for the design of the sampling methodology, training the team of community members with whom I worked to collect the field data, evaluating the sampling data, analyzing the data, interpreting the results and writing the manuscript. Again, discussions with Dr. Potvin contributed to each stage of the process, and feedback on initial drafts influenced the final paper.

Acknowledgements

I would like to thank my co-supervisors, Dr. William Laurance and Dr. Catherine Potvin, for their ongoing support and for sharing with me their passions for tropical ecology as well as their experience and philosophies surrounding tropical conservation and development issues.

Many people contributed to the studies presented in this thesis. Discussions with Dr. Henrique Nascimento throughout my time at STRI contributed to the work presented in both Chapters 2 and 3. Dr. Petra Tschakert's ideas and experience in the field contributed to the study presented in Chapter 3. I am grateful to Benjamin Gilbert for the time, energy, and ideas he contributed to the work presented herein. My work in Ipetí would not have been possible without the support of the members of the Organisación de la Unidad y Desarrollo de Ipetí-Emberá; in particular, Bonarge Pacheco, Omaira Casama, Jeremia Cansari, Pablo Guainora and Ultimilio Cabrera. The members of the carbon inventory team, Jaime Caisama, Leonel Caisama, Charianito Cansari, Juan Casama, Nesar Dumasa, and Villalaz Guainora collected the inventory data and shared their knowledge of the plants, animals and politics of Eastern Panama during our time in the field. I am grateful to Oris Acevedo, Dr. Hector Barrios, Mark Brooks, Dr. Richard Condit, Dr. Luis Cubillo, Dr. Quentin Gall, Dr. Nelida Gomez, Natalia Molina, Paulo Monteiro Brando, Dr. Tim Moore, Jackie Ngai, Rolando Perez, the Ruiz family, Nilka Tejeira, and Raineldo Urriola for assistance in many forms at different stages of the project. Finally, I would like to thank my family for their support, encouragement and great senses of humour. The National Science and Engineering Research Council of Canada and the Canadian Forest Service provided me with scholarship support.

List of abbreviations

AGB	Above-ground biomass
ANAM	Autoridad Nacional del Ambiente, Panamá
BD	Basal diameter
C	Carbon
CDM	Clean Development Mechanism
CO_2	Carbon dioxide
CTFS	Center for Tropical Forest Studies
DBH	Diameter at breast height
GIS	Geographical Information System
GPS	Geographical Positioning System
FAO	Food and Agriculture Organization
IPCC	Intergovernmental Panel on Climate Change
LOI	Loss on ignition
N	Nitrogen
SOC	Soil organic matter
TC	Tierra Colectiva de Ipetí-Emberá (collective lands of Ipetí-Emberá)
UNFCCC	United Nations Framework Convention on Climate Change

Abstract

Land-use change, and in particular tropical deforestation, is the leading cause of species extinctions globally, and is the second most important source of CO_2 emissions after fossil fuel combustion. I examine two policy-relevant questions that relate to tropical deforestation and land use change: (1) At regional scales, what biophysical and infrastructure-related factors are associated with deforestation? and (2) At a local scale, what are some of the impacts of land use change on above- and below-ground carbon stocks and on tree-species richness? The first question was examined for the Brazilian Amazon through spatially-explicit correlation analyses of deforestation and a series of predictor variables that included highways and roads, annual rainfall, dry season length, soil characteristics, site accessibility, and population density. The proximity of a site to roads and highways was the strongest predictor of deforestation, with more accessible sites more likely to be deforested. Dry season length was also a strong, positive predictor of deforestation. The results suggest that current plans to expand road infrastructure in Amazonia will have a significant impact on the forests of the areas transected.

The second question was examined in the context of a 3,198 ha area in Eastern Panama that is managed collectively by an Indigenous Emberá community. The above- and below-ground carbon stocks of forests, agroforests, and pastures in the collective lands were quantified, and the average number of tree species per ha in each land use type determined. The average C stocks (including roots and soil organic carbon to a depth of 40 cm) were 255 Mg C ha^{-1} for forests, 127 Mg C ha^{-1} for agroforests, and 45 Mg C ha^{-1} for pastures. The number of native tree species per hectare was about two times higher in forests than in agroforests, and five times higher in forests than in pastures. The

study revealed the potential for externally funded projects to provide incentives that would have concurrent benefits for carbon and biodiversity conservation.

Abstract

Les changements d'utilisation des terres, et en particulier la déforestation tropicale, est la cause principale au niveau global de l'extinction d'espèces, et la seconde cause d'émissions de CO_2 après la combustion de combustibles fossiles. Cette thèse considère deux questions d'importance associées à la déforestation et l'usage de terre tropicale. Premièrement, a l'échelle régionale, quels facteurs biophysiques et infrastructurels sont associés avec la déforestation? Deuxièmement, à l'échelle locale, quels sont les impacts du changement d'utilisation des terres sur les quantités de carbone au-dessus et en dessous du sol et sur la richesse d'espèces d'arbres? La première question a été examinée dans le contexte de l'Amazone Brésilien grâce à des analyses spatiales qui tentaient d'établir les relations entre la déforestation et une variété d'autres facteurs incluant les autoroutes, les routes non asphaltées, la précipitation annuelle, la durée de la saison sèche, l'accessibilité des sites, les caractéristiques des sols, et la densité des populations humaines environnantes. L'accessibilité d'un site par routes et autoroutes était la variable prédictive la plus importante; les sites plus accessibles avaient une plus grande probabilité d'être déforestés. La durée de la saison sèche est également fortement et positivement reliée à la déforestation. L'agrandissement du réseau routier en Amazonie Brésilienne pourrait avoir de fortes conséquences quant au couvert forestier.

La deuxième question a été examinée dans le contexte d'un territoire de 3,198 ha géré collectivement par une communauté indigène Emberá dans l'est de la République du

Panama. La quantité de carbone au-dessus et en dessous du sol a été mesurée dans des parcelles de forêts, d'agroforêts, et de pâturages. Le nombre d'espèce d'arbre par hectare dans chaque écosystème a aussi été examiné. Les quantités moyennes de carbone dans chaque système étaient (incluant racines et matière organique du sol à un profondeur de 40 cm): 255 Mg C ha^{-1} pour forêts, 127 Mg C ha^{-1} pour agroforêts, et 45 Mg C ha^{-1} pour pâturages. Le nombre d'espèces indigènes par hectare était à peu près deux fois plus élevé en forêts qu'en agroforêts, et cinq fois plus élevé en forêts qu'en pâturages. L'étude révèle un fort potentiel pour développer des projets de puits de carbone dans le cadre du protocole de Kyoto. De tels projets sauraient encourager des activités ayant des bénéfices simultanés pour le séquestration de carbone et la conservation de la biodiversité.

Chapter 1

Introduction

Land-use change, and in particular tropical deforestation, is the leading cause of species extinctions globally (FAO 2001), and is the second most important source of CO_2 emissions after fossil fuel combustion (IPCC 2000). During the 1990s, an average of almost 15 million ha of forest were lost a year, mainly in the tropics (FAO 2001). In addition to effects on biodiversity and carbon storage, deforestation alters hydrological cycles, is associated with soil erosion and degradation, and affects the forest resources available to adjacent communities (Aylward et al. 1998; Byron and Arnold 1999). This thesis examines two questions that stem from tropical deforestation and land use change. First, at regional scales, what biophysical and infrastructural factors are associated with deforestation? Second, at a local scale, what are some of the impacts of land use change on above- and below-ground carbon stocks and on tree species richness?

The causes and correlates of deforestation

In contrast to much of the deforestation that is currently taking place in temperate regions, and certainly in Canada, tropical forests are generally cleared to establish agriculture or alternative land uses, and not for their timber. A conceptual model of the factors influencing deforestation is presented in figure 1. At the highest level are global- to regional-scale economic and development policies. These macroeconomic policies influence a set of variables that are tangible to the individuals or companies who make decisions on how to allocate land to alternative uses (e.g. small-scale farmers, cattle ranchers, and resource-extracting companies). These more tangible variables include institutions (such as land-clearing and land-ownership laws), markets, and infrastructure that allows access to forested land. Agents' decisions on how to manage their land are influenced by these variables, by the biophysical characteristics of the land, and by the agents' own experiences, values, needs and resources (Angelsen and Kaimowitz 1999; and Geist and Lambin 2001).

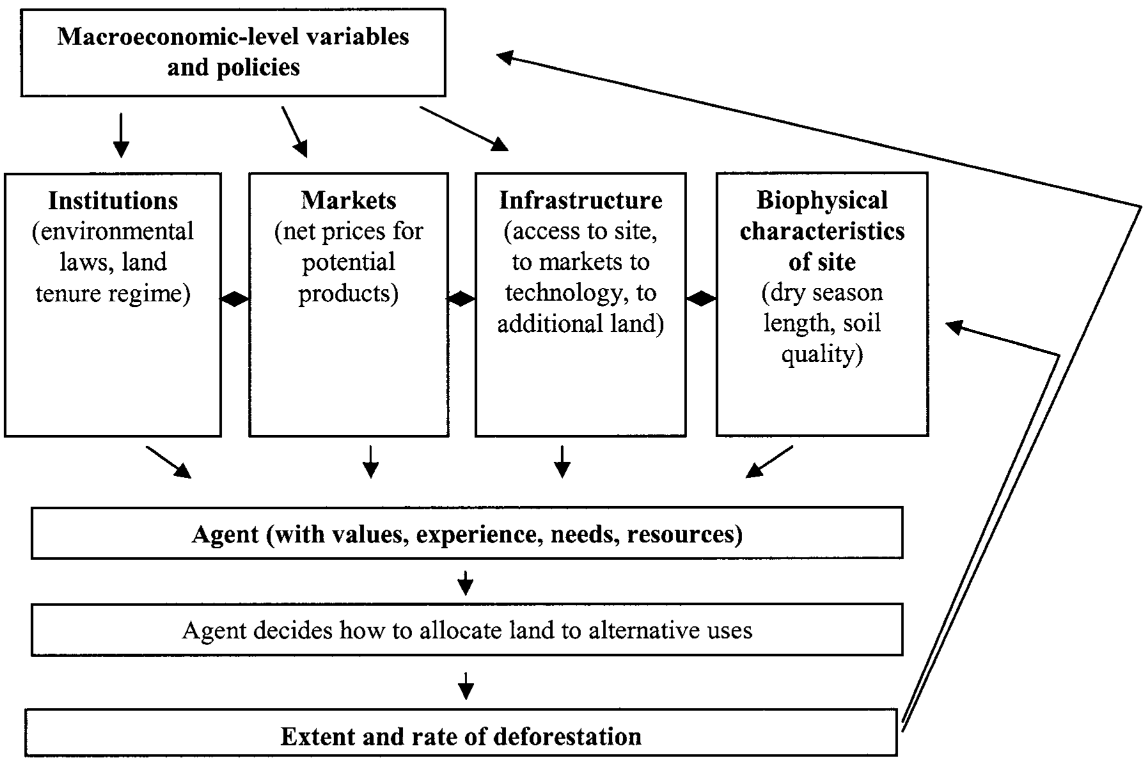

Figure 1. Macroeconomic-level policies influence institutions, markets and access to sites. These external factors are perceived by the agent, who considers these factors, the biophysical characteristics of the land, and his or her own values, resources, needs and experience in making a decision on how to allocate available land to alternative uses. (Variables from Angelsen and Kaimowitz (1999) and Geist and Lambin (2001)).

This model provides a framework with which to consider the debates surrounding the determinants of deforestation. Traditional arguments blame deforestation on population growth and poverty, and place little to no responsibility on governments and macroeconomic policies. However, if deforestation is considered from the point of view of the agent carrying out the deforestation, and if we accept that agents act in response to perceived opportunities, then the potential roles of macroeconomic policies in determining those opportunities are difficult to deny. In general, case studies of deforestation provide overwhelming support for the role of macroeconomic policies in influencing deforestation (Angelsen and Kaimowitz 1999; Geist and Lambin 2001; Lambin et al. 2001). For example, credit provided to cattle ranchers in Amazonia in the 1970s and subsidies that continue to be provided to Amazonian soy producers are two forms of top-down market manipulation that have been shown to have made important contributions to deforestation in the Brazilian Amazon (Moran 1993; Fearnside 2001).At a global level, devaluation of the Real is currently having a big impact on Amazonian deforestation by making Brazilian soy and beef exports much more competitive internationally, and thus providing incentives to cattle ranchers and soy farmers to expand export-oriented production (Kaimowitz et al. 2004). These large-scale producers tend to buy already-cleared land from small-scale farmers, pushing the smallholders to new frontiers (Laurance et al. 2004). Institutions have similarly been shown to be important drivers of deforestation. In the Brazilian Amazon, clearing land is considered an "improvement" by law and is a first step in the process of land titlement (Hecht 1990). One of the most important ways that governments can influence land use is by developing infrastructure that improves access to previously difficult-to-reach areas. Not only do roads allow settlers access to new areas for agriculture, but opening new frontiers reduces the pressure on landowners in 'old' frontiers to intensify their land use or to ensure the sustainability of their land use over time (Geist and Lambin 2001). In Panama, the extension of the Interamericana highway into forested areas of the Darien in the 1980s has provided cattle ranchers from the interior of Panama with access to new pasture lands; there are virtually no forests left to clear in the dry Pacific coastal zones of the Panamanian interior, and the productivity of much of the extant pasture has been exhausted (Heckadon-Moreno 1997).

Feedbacks and synergies among the different levels and variables of the model should also be emphasised. These interactions may be planned (e.g. colonization schemes and special credit programs may be used to lure settlers into newly accessible areas), or they may arise as a result of supply-and-demand forces (e.g. as forested land becomes more scarce, the market price for forest products will rise).

In Chapter 2 of this thesis I examine the relative importance of infrastructure and site biophysical characteristics to deforestation in the Brazilian Amazon. In particular, I use spatially-explicit analyses to relate deforestation to a series of infrastructural, demographic and biophysical predictor variables. The use of large-scale satellite imagery and census-based demographic data are relevant to the scales at which planning is currently taking place in Brazil (Laurance et al. 2001). I use the results from this analysis to predict the probable effects of government plans to expand road infrastructure through previously remote areas of the Amazon. In addition, I consider arguments that both biophysical characteristics of these 'new frontiers', and stricter environmental laws will prevent patterns of past deforestation from being repeated.

Deforestation, land use and environmental services

As rates of deforestation continue to accelerate in many parts of the world, the potential for market mechanisms to increase the perceived values of forests to landowners and other agents of deforestation is receiving increasing attention from economists and ecologists alike (Pearce 1996; Costanza et al. 1997; Balmford et al. 2001). Market-based approaches to controlling deforestation argue that until markets capture the value of forests to groups other than the land owner, forests will be undervalued and there will be few incentives to conserve them (Pagiola et al. 2002). These groups might be downstream communities that benefit from flood control and reduced sedimentation, or the global community that benefits from avoided carbon emissions and biodiversity loss. Klooster and Masera (2000) suggest that this is the only realistic means for conserving forests in inhabited areas, where forests are used intensively for wood, fuel, timber, and agriculture, often "clandestinely and without coordination."

Costa Rica was one of the first countries to institute such an incentive system. Since 1997, the country's System of Payments for Environmental Services (Pagos por Servicios Ambientales, PSA) has paid forest owners for the green house gas-mitigation, hydrological services, biodiversity, and scenic beauty that their forests provide. Forest owners are required to develop a sustainable forest management plan which must be certified by a licensed forester. The land owner then receives about US$40 per hectare per year (Pagiola 2002). The Clean Development Mechanism (CDM) of the Kyoto Protocol provides a framework establishing a payment system for carbon services at the global scale. The CDM allows developed countries that have committed to lower their carbon emissions to earn 'emissions credits' by enabling projects in developing countries that either sequester carbon or produce clean energy. The projects must also contribute to the sustainable development of the host country (UNFCCC 2001). Although the Kyoto Protocol has yet to be ratified, carbon trading systems modeled on the CDM are already active (reviewed in Grace, 2004).

The growth in markets for environmental services will demand that the methods for quantifying environmental services keep pace. Once developed, these methods will likely become as routine as those currently used to assess wood volume for logging operations. However, a lot of debate surrounding methodologies for quantifying environmental services remains (Pagiola et al. 2002).

A current challenge lies in transferring methods and "lessons learned" from academic-oriented studies to applied attempts to establish environmental services-based projects. In the case of carbon projects, there have been few attempts to develop standard methodologies for the measurement of carbon stocks, despite the active debate in the academic literature surrounding the error introduced by alternative measurement methodologies at scales ranging from landscapes to individual trees (Clark et al. 2001; Chave et al. 2003; Chave et al. 2004; though see MacDicken 1997). Studies of carbon stocks to date tend to provide either very precise estimates of the carbon stocks of one land use type in one region (e.g. Chave et al. 2004; Chambers et al. 2001) or estimates of carbon from a number of adjacent land use types that could provide the basis for more applied projects, but that are published without estimates of the error or uncertainty in the estimates (e.g. Fujisaka et al. 1998; Kotto-Same et al. 1997).

In Chapter 3 I quantify the carbon stocks in three land use types in Eastern Panama. I attempt to develop a sampling method that is repeatable, allows for easy comparisons among land use types, and incorporates local ecological knowledge such that it can be adopted by local communities. Finally, I examine the role of environment and management factors in determining the above- and below-ground carbon stocks of sites in the three land use types. I examine the results in terms of their implications for a carbon management project in the community

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Chapter 2*

The future of deforestation in the Brazilian Amazon

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Abstract

Concern about the future of Amazonian forests is growing as both the extent and rate of primary forest destruction increase. We combine spatial information on various biophysical, demographic and infrastructural factors in the Brazilian Amazon with satellite data on deforestation to evaluate the relative importance of each factor to deforestation in the region. We assess the sensitivity of results to alternative sampling methodologies, and compare our results to those of previous empirical studies of Amazonian deforestation. Our findings, in concert with those of previous studies, send a clear message to planners: both paved and unpaved roads are key drivers of the deforestation process. Proximity to previous clearing, high population densities, low annual rainfall, and long dry seasons also increase the likelihood that a site will be deforested; however, roads are consistently important and are the factors most amenable to policymaking. We argue that there is ample evidence to justify a fundamental change in current Amazonian development priorities if large-scale losses of forests and environmental services are to be avoided.

Keywords

Amazon, Brazil, deforestation, highways, land use, population density, roads, spatially-explicit analysis

1. Introduction

Almost 70% of the Amazon basin falls within Brazil's borders and the country sustains 40% of the world's remaining tropical forests. Within Brazil, the "Legal Amazon"[1] region covers 58% of the national territory and shares borders with all eight other Amazonian countries: Bolivia, Peru, Ecuador, Columbia, Venezuela, Guyana, Suriname and French Guyana. The region's geopolitical position, size and low population

[1] The "Legal Amazon" includes the states of Amapá, Amazonas, Rondônia, Roraima, Pará, Maranhão (west of 44°W), Tocantins, Goiás (north of 13°S), and Mato Grosso.

density have meant that it has long been seen as “strategically vulnerable and economically underutilized” by federal planners [1]. Indeed, the Legal Amazon still houses only 11% of Brazil’s population, and in 1999 it contributed just 4% of the country’s GDP [1]. However, the forests of the Amazon basin also provide environmental services that are important both locally and globally including the conservation of biodiversity, carbon storage, and the regulation of regional hydrological cycles, among others [2]. Concern about the future of Amazonian forests is growing as both the total extent and rate of primary-forest destruction increase (fig. 1).

Several recent studies of land use change in the Brazilian Amazon have used empirical methods to describe the relationships among deforestation and its ‘driver’ variables, be they biophysical, infrastructural or demographic factors. Although our growing understanding of these relationships could inform policy and decision-making processes, federal-level planning for the Amazon region continues to emphasise projects that will maximise foreign earnings through benefits to export-oriented industries, with little regard to the impacts of planned projects on the forest landscape. In this paper we first describe the historical development policies and land uses that have contributed to deforestation in the Brazilian Amazon. We then present the findings of a new, spatially-explicit analysis of the predictors of Amazonian deforestation and compare these findings to previous empirical studies of Amazonian deforestation. Finally, in light of current trends in factors that have been shown to be strongly related to forest clearing, we discuss the future of deforestation in the region.

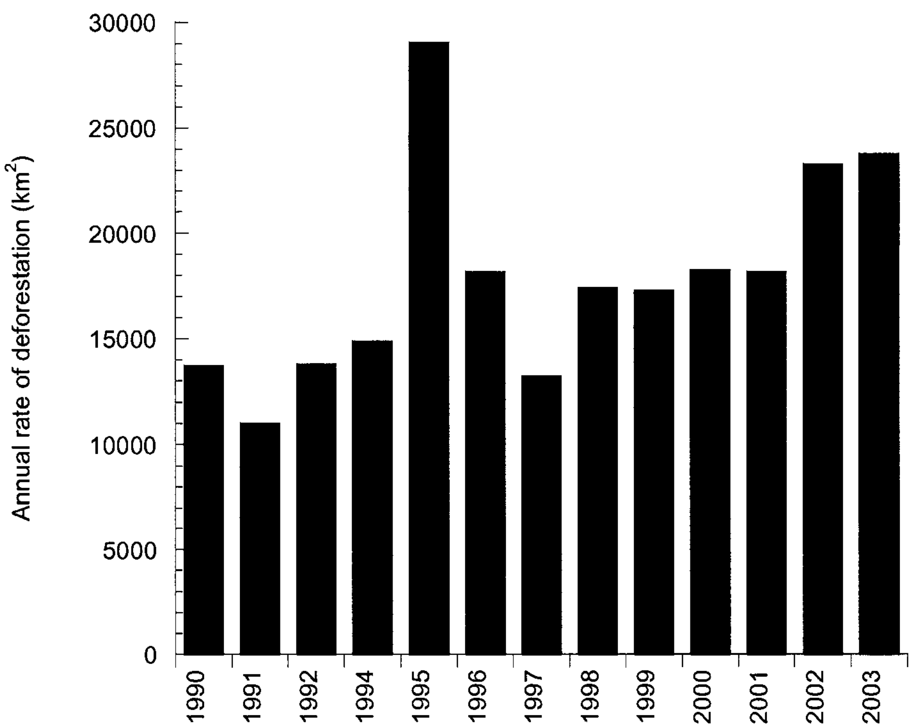

Figure 1. Annual rates of deforestation in the Legal Amazon from 1990-2002. Deforestation was not measured in 1993, and the mean annual rate of change from August 1992- August 1994 is therefore presented for the 1994 year.

2. Historical development trends in the Brazilian Amazon

Approximately 4 million km^2 of the 5 million km^2 Legal Amazon region of Brazil were forested at the beginning of the 20th century, with the remaining areas covered by naturally occurring savannah shrub lands (*cerrado*) and savannah grasslands (*campos naturais*). Prior to the early 1960s access to the Amazon was severely restricted and aside from limited clearing along rivers the forest remained essentially intact. Construction of the first road through the region, the Belém-Brasilía highway, began in 1958 with the goal of integrating western and northern states with the rest of the country [3]. The initiation of the Cuiabá-Porto Velho (BR-364) highway followed in 1968 to provide access to the southern portion of the Amazon. These first two highways – the only federal highways in the Legal Amazon to be paved and therefore passable year-round before the late 1990s – are at the heart of the "arc of deforestation," which to date is the focal region of deforestation in the Brazilian Amazon [4, 5] (fig. 2). Several other mostly unpaved highways that have been important in the historical deforestation of the Amazon include the Transamazon (BR-230), which runs west to east from Lábrea through Marabá, the BR-163, which runs south to north from Cuiabá to Santarém, and the BR-174, which runs south to north from Manaus through Boa Vista (fig. 2).

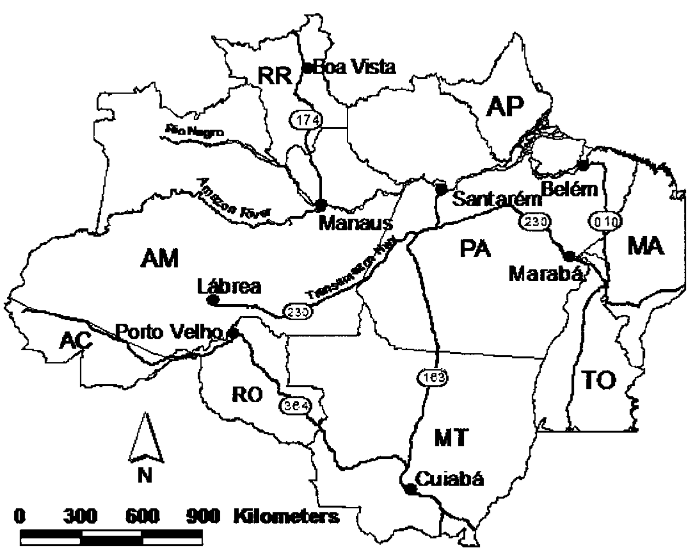

Figure 2. Boundaries of the Legal Amazon and its states (AC = Acre; AM = Amazônas; AP = Amapá; MA = Maranhão; MT = Mato Grosso; PA = Pará; RO = Rondônia; RR = Roraima; TO = Tocantins), and cities and highways mentioned in the text.

In addition to improving transport infrastructure, the government used various incentives to encourage colonisation and the development of intensive economic activities in the region throughout the 1964-1985 military dictatorship period [1]. These incentives were overwhelmingly directed at extensive cattle ranching projects (631 of 950 projects approved for funding between 1966 and 1985) [6]. Large-scale mining, timber extraction and hydroelectric energy projects were also undertaken. In focusing on intensive economic activities the government proposed to produce revenues that would service Brazil's foreign debt and finance further development [1]. Aside from a few localized government settlement programs (particularly along the Transamazon highway in the early 1970s and in the state of Rondônia from the mid-1970s onwards [7]), colonization was generally unorganised and was expected to occur on its own around large projects [1]. In the case of the Belém-Brasilía highway, two million people settled along the highway in its first 20 years, mainly in the state of Pará [3].

Calls for the conservation of Brazil's rainforests began to emerge in the mid-1970s with the publication of the first estimates of the extent of Amazonian deforestation. By the late 1980s, international interest in the conservation of the Amazon had begun to affect Brazil's ability to attract foreign investment and financial support for large projects [3, 8]. The Brazilian government's increasing preoccupation with its environmental image is reflected in its Amazonian policy beginning in 1985. Incentives to cattle ranchers were formally withdrawn through a series of decisions and decrees in 1985, 1989, and 1991[2] [9, 8, 3]. In 1988, as part of the new constitution, the destruction of Amazon and Atlantic rainforests became a crime under the penal code, although little

[2] Though see Fearnside [18]: "contrary to popular belief, many ranchers still receive fiscal incentives because the June 25, 1991 decree (no. 153) on incentives only suspended granting *new* incentives, rather than revoking old (already approved) ones".

attention was given to the enforcement of these laws [9]. In 1989, Brazil volunteered to host the 1992 UN Conference on Environment and Development (UNCED), and in 1993 the Pilot Program to conserve the Amazon (PP-G7) was launched by the Group of Seven industrialised countries (Canada, France, Germany, Italy, Japan, United Kingdom, United States of America) in cooperation with Brazil. Germany was the largest contributor to the PP-G7's budget of $250 million, which was to finance such initiatives as the demarcation of indigenous territories and extractive reserves, the strengthening of environmental institutions and local governments, and NGO demonstration projects [10].

The launch of the PP-G7 coincided with the general recession of the late 1980s that alone reduced development initiatives and therefore deforestation throughout Brazil [9, 3]. President Fernando Collor de Melo, who had led the charge to present a positive environmental image of Brazil, resigned in late 1992 under threat of impeachment for corruption [8]. Following the UNCED Rio Summit, Amazonia disappeared from both the international and the Brazilian press, and Brazil's politics began to shift back toward promoting the interests of military, mining, construction and agricultural groups. Nationalist sentiments also contributed to this shift [8].

3. Land use and deforestation

In terms of land use activities, cattle ranching and small-scale farming have historically played the most significant role in the clearing of Amazonian forest. In addition, the importance of soy farming as a land-demanding economic activity has grown dramatically in the last ten years [11, 12]. Each of these activities tends to be strongly associated with agricultural establishments of particular sizes; data from the most recent national agricultural census reveals both the highly unequal distribution of land in the legal Amazon and the disproportionate contribution of cattle ranching and soy farming to deforestation (table 1). For example, properties greater than 2000 ha in size, which tend to be cattle ranches or soy farms, constitute only 1% of all agricultural establishments in the nine Amazonian states but control 46.8% of all land converted from forest or *cerrado* to agriculture. In contrast, subsistence farms of less than 20 ha constitute over 50% of establishments in Amazonia but control only 1.5% of land converted to agriculture [13].

Table 1. Size of agricultural establishments in the nine Amazonian states and their principal agricultural products based on Brazil's 1995/1996 agricultural census. Note that 2% of establishments could not be classified from the census data (IBGE 2001; Chomitz and Thomas 2001).

Size of establishment	Percent of all establishments	Percent of all land converted from forest or cerrado to agriculture controlled by establishments of this size	Principal agricultural products
< 20 ha	54	2	Subsistence (manioc, rice)
20-100 ha	28	Not available	Subsistence, some cash products (manioc, bananas, milk)
100-2000 ha	15	Not available	Cattle, soybean
> 2000 ha	1	47	Cattle, soybean

The historical role of cattle ranching in Amazonian deforestation is partly a result of the favourable incentives received by cattle ranchers throughout the 1965-1985 period. Economic analyses have shown that where credit was available, converting forest to pasture was more profitable than the sustainable use of already-cleared lands [3, 14]. However, even at the height of the government incentives programs in 1975, over 45% of clearing along the Belém-Brasilía highway was in agricultural establishments – almost all large ranches – that received no government subsidies [7]. In part, this reflects the attractiveness of cattle ranching to Amazonian farmers: cattle are a highly liquid investment that can be readily sold if necessary; there are strong local markets for beef throughout Brazil; cattle can be brought to the market on foot and therefore do not require truck-grade roads; sales of cattle can be delayed without incurring major losses; cattle ranching is not labour-intensive; cattle produce milk, skins, manure, offspring and meat and are less vulnerable to annual variation in weather than crops; and cattle ranching has traditionally been regarded as a prestigious activity in Brazilian society [7, 15, 16].

Today Brazil's cattle herd is the largest in the world [12]. However, numerous studies have emphasized that large landholders in Amazonia are generally less interested in raising cattle than in securing their land tenure. Under Brazilian legislation, clearing land for pasture is considered an "effective use" of land and is a first step towards securing land ownership [17]. Securing ownership is critical to both land speculators and large landholders because of the threats of invasion by landless peasants or of expropriation by a land redistribution program. Cleared land is also worth 5-10 times more than forested land, and clearing is therefore well worthwhile to the owner whose ultimate goal is resale [15]. The strong performance of land prices in the face of Brazil's high rates of inflation in the 1970s and 1980s and the fact that capital gains taxes are almost never collected has meant that land speculation has long been popular in Amazonia [3, 18]. The cheapest and most efficient way of maintaining cleared land is by cattle grazing, and the ubiquity of cattle operations with very low stocking densities in Amazonia suggests that maintaining land cleared is indeed a prime motivation for much of the cattle ranching that is underway in the region [19, 15].

After cattle ranchers, small farmers have played the most significant role in the clearing of Amazonian forest. Most of the original small-farmer immigrants to Amazonia came from drought-stricken northeastern states or from south-central Brazil where increasing industrialization of farming was leading to land concentration and to the expulsion of small-holders to new frontiers. In the Amazon, the initial land claim by small farmers is accompanied by farm creation using slash and burn. The extent and rate of clearing are determined by labour supply and capital, and the process of farm establishment may span a decade or more [20, 21]. The average small farmer in Amazonia clears 1 ha of forest per year [22]. A plot can generally support annual crops for 2-3 years, after which soils are exhausted and new areas cleared. Old fields are left fallow or converted to pasture. In view of the expense of fertilizers, the shortage of labour in Amazonia, and the abundance of inexpensive forestland, several studies have argued that slash-and-burn is by far the most economical means for farmers to improve the fertility of the soils [7, 23]. Unfortunately, the fallow periods are rarely long enough to allow soils to recuperate fully, and the system is therefore not sustainable [7].

More recently, a long growing season, the development of new cultivars, ample agricultural financing and cheap land prices have fostered the rapid expansion of the soy industry in Brazil and the country is currently the second global producer of soybeans after the United States. Historically soy producers have been concentrated in southern and central Brazil and in savannah areas of the Legal Amazon. However, as prices for soybeans continue to rise, soy producers are pushing northwards into forested areas of the Amazon, particularly along the recently paved portions of the Cuiabá-Santarém (BR-163) highway [11, 12]. Soy farmers generally buy already-cleared land from small farmers, displacing the small-holders to cities or to new frontiers; in the latter case, the process of farm establishment is reinitiated [24]. Because soy farming depends heavily on agricultural inputs and machinery, it is almost solely the domain of wealthy agribusinessmen, and soy farming has been associated with extreme income concentration wherever it has spread in Latin America [25]. As the soy industry is now a major source of foreign currency for Brazil, the needs of soy farmers have been used to justify many of the controversial transport infrastructure projects that are currently underway in Amazonia [11]. These are discussed further in Section 5.

Currently about one-third of the Legal Amazon is classified as protected areas or indigenous lands [26, 27]. Indigenous reserves represent 76% of this area and encompass 22.5% of the Amazonian biome. Totally protected areas that do not overlap with indigenous areas account for only 3.6% of the Amazonian biome, and sustainable use areas (most of which are national forests and are subject to industrial logging) represent 9.0% [27, 26]. Studies have shown that although status as indigenous land or protected area does provide protection against outright deforestation, these areas are the centre of much of the legal and illegal logging activity that is currently taking place in Amazonia [28]. As they are increasingly surrounded by roads and deforestation, indigenous lands and protected areas are also vulnerable to poaching and degradation by runaway fires [29, 23].

4. Empirical measures of deforestation in the Brazilian Amazon

4.1 Background

The Brazilian National Institute for Space Research (INPE) has produced annual, satellite-based estimates of deforestation since 1989, with the exception of 1993. Types of clearings that contribute to deforestation estimates include pasture, agricultural plots, areas of gold and other mining activity, areas flooded as hydroelectric reservoirs, roads and urban areas. Activities such as selective logging and surface fires that may significantly thin the forest canopy but which do not destroy it entirely are not included in INPE estimates of deforestation [30].

Two trends can be observed in the chart of annual rates of deforestation of 1990-2003 (fig. 1). The first is that the annual variation in deforestation is large, and the second is that the annual rate of deforestation is accelerating (r_s=0.66; p=0.014). The first trend probably reflects the year-to-year variation in factors that affect the ability of land-users to clear forest, such as disposable income (which in turn might reflect factors such as the state of the national economy and inflation rates) and the length of the dry season (e.g. the long dry seasons of El Niño years facilitate extensive land-clearing with fire) [31, 32, 33, 34, 35]. For example, the dramatic jump in deforestation in 1995 has been attributed to the increase in available investment funds following the federal economic reforms in July 1994 that stabilized the Brazilian currency [36]. The sensitivity of the Amazon-wide deforestation rate to annual variation in factors such as export prices will further depend on which economic activities are dominant in the region. For instance, whereas subsistence households are generally not very sensitive to market fluctuations [37], cattle farmers' disposable income (and therefore ability to clear land) is dependent on local beef markets. However, beef markets in Brazil are relatively stable in comparison to global commodity markets such as the soy bean market on which a growing number of Amazonian landowners depend [11]. The forces that drive annual variation are not discussed further in this paper; however, the dramatic jumps in deforestation in 1995, 2002 and 2003 reflect their importance.

The second trend, the acceleration of annual rates of deforestation from 1990-2003, is driven by factors at local, national and international levels. The drivers are in some cases difficult to isolate as single, quantifiable variables. Often, however, the factors driving deforestation are measured by government censuses, independent field studies, or satellite-based remote sensing projects. When data for these factors are available, their relationships to deforestation can be assessed empirically.

A number of methods have been used to study the relative contribution of different factors to deforestation; Angelsen and Kaimowitz [37] provide a thorough review of deforestation studies and methodologies. Traditionally, empirical studies have used deforestation estimates based on agricultural census data or on reports of land use by different government institutions (e.g. [38, 13, 16]). The advent of remote sensing and Geographic Information Systems (GIS) has allowed researchers to link census-based data on demographics and socio-economics to satellite-based data on land use change (e.g. [39, 40, 41, 15]). Factors that are often included in such analyses include density of or proximity to paved roads, unpaved roads and rivers; proximity to markets or major population centres; presence of protected areas; climate; and edaphic characteristics. In the following section we present the results of a spatially-explicit analysis of the relationships of deforestation to several of its most-commonly cited driver variables. This study closely follows a study by Laurance and colleagues [41] that examined these relationships for a random sample of 120 sites in the Legal Amazon. However, in the following section we tackle two methodological issues that were not taken into account in the original study. First, we stratify our sampling not only on deforestation intensity, as was done in the original study, but also on each explanatory variable in turn. This allows us to determine whether the relationship of deforestation to each driver variable changes when sampling is stratified to include a full range of values for the driver variable. Second, for each stratified variable, we draw and analyse ten random samples of 120 sites. This allows us to assess the degree to which our results are influenced by the random selection of sites. Our approach allows us to draw strong conclusions regarding the relative importance of different driver variables to deforestation.

4.2 Methodology

The data analysed in this study were developed by Laurance and colleagues [41]. GIS software (ArcView GIS 3.2) was used to develop compatible spatial coverages for the distributions of deforestation, population density, roads, rivers, soils, annual rainfall and dry season length for the entire Legal Amazon region. A sampling grid of 50 km by 50 km grid cells was overlaid on each distribution, and data on deforestation and all predictor variables were extracted for each of the 1867 grid cells. Data sources and methods are consistent with those described in [41], with two exceptions. First, although our soil data is drawn from the same map of the Brazilian Agricultural Research System (EMBRAPA), we reclassified soil units to place much more weight on physical restrictions to soil use, as chemical restrictions are more easily corrected through the use of chemical fertilizers; the soil classification scheme is summarized in Appendix 1. Second, we withdrew proximity to navigable rivers as a predictor variable from this study, as the original study revealed that the grain of the analysis was too coarse to detect clearing along rivers, and the correlation analyses were therefore unable to capture the relationship of deforestation to river access.

In order to minimize the effects of spatial autocorrelation among response and explanatory variables, the original study examined relationships among deforestation and predictor variables for a random sample of 120 of the 1867 grid cells stratified on deforestation intensity. In this study, we replicate this methodology but use two alternative sampling regimes to thoroughly examine the relationship of each predictor variable to deforestation. In the first case, we stratify sampling on deforestation intensity as was done in the original study. In the second case, we stratify sampling on each predictor variable in turn, such that the full range of values of each predictor is present in the samples of 120 cells that are analysed to determine the relationship of the particular predictor to deforestation. The original study examined only one deforestation-stratified sample of 120 grid cells. In contrast, for each stratification method, we drew and analysed ten random samples of 120 grid cells. We performed regression analyses for each random sample, with the percent deforestation arcsine-square root transformed and certain independent variables (distance to paved and unpaved roads and density of rural and urban populations) log transformed to meet regression model assumptions. This allowed us to assess the sampling error, to ensure that any correlation we detected was not spurious, and to calculate a mean coefficient of determination (r^2) for each predictor. The coefficient of determination describes the percentage of variation in the response variable that is explained by each predictor. We did not perform multiple regression analysis because of the strong colinearity between some independent variables. The coefficient of determination reported thus represents the total potential variation explained, without accounting for other variables.

4.3 Results

The correlations of all predictors to deforestation were significant at the α=0.05 level for every sample drawn and for every predictor variable, with the exception of the soil variables (fig. 3). Soil fertility was significantly correlated to deforestation in just 1/10 and 0/10 of the deforestation-stratified and predictor-stratified samples respectively, and the soil waterlogging variable in only 2/10 and 3/10 samples respectively.

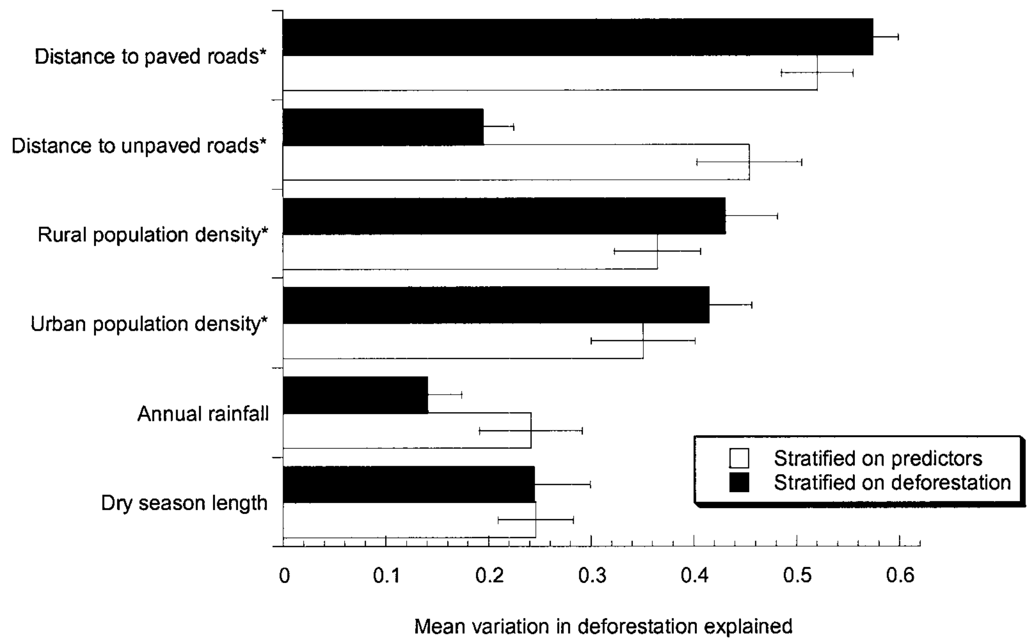

Figure 3. Mean variation (and 95% confidence intervals) in deforestation explained by each predictor variable when sampling is stratified on deforestation intensity (black bars) and when sampling is stratified on each predictor variable in turn (white bars). Predictors with a '*' were log transformed to meet assumptions of regression analysis.

When sampling was stratified on deforestation intensity, paved roads were the best predictor of deforestation, with sites closer to paved roads more likely to be deforested. Paved roads explained 38% more variation in deforestation intensity than did unpaved roads. Rural and urban population densities were also strongly correlated with deforestation, with more deforestation in more densely populated areas. Annual rainfall and dry season length were less important as predictors than paved roads or population density; however sites with less annual rainfall and longer dry seasons were consistently more likely to be deforested than those with more annual rainfall and shorter dry seasons.

When sampling was stratified on predictor variables, paved roads again explained more variation in deforestation than did any other predictor variable, with the mean coefficient of determination (r^2) within 5% of that for deforestation-stratified samples (fig. 3). In contrast, the relationship of unpaved roads to deforestation changed dramatically, with roads explaining over 25% more variation in deforestation when sampling was stratified to include a full range of road values. The strength of the correlation of urban and rural population densities was reduced when sampling was stratified to include a full range of population values (by 6.5% and 10% for rural and urban population densities, respectively). Annual rainfall explained 10% more variation in deforestation intensity when sampling was stratified on annual rainfall, however the correlation of dry season length to deforestation changed very little.

All of the drivers examined therefore appear to be significant predictors of deforestation at the scale of our study, with the exception of the soil variables. The relationships of the explanatory variables to deforestation under the two stratification methods were relatively consistent, except in the case of unpaved roads, suggesting that this factor in particular merits further investigation.

4.4 Discussion

We found that areas that are accessible by paved or unpaved road, have high population densities, and have relatively low annual rainfall and long dry seasons are more likely to be deforested than are areas with opposite features. Our findings are consistent with those of previous empirical studies that have examined the predictors of deforestation on an Amazon-wide scale [38, 40, 13, 41] (table 2).

Table 2. Results of five Amazon-wide studies of the predictors of deforestation.

Study	Reis and Margulis 1991	Pfaff 1997	Chomitz and Thomas 2001	Laurance et al. 2002	This study
Response variable	Extent of deforestation (census/agency reports)	Cleared land density (satellite)	Proportion of area under agriculture (census)	Deforestation (satellite)	Deforestation (satellite)
Extent of study	Legal Amazon	Legal Amazon	Legal Amazon	Legal Amazon	Legal Amazon
Unit of analysis	Municipality	Municipality	Census tract	50 km by 50 km cell	50 km by 50 km cell
Year of response variable data	1983-1987 depending on state	1988	1996	1998	1998
Proximity to or density of roads	+	+	+	+	+
Density or extent of previously cleared land*	+	+	+	not tested	not tested
Proximity to or density of rivers**	not tested	-	-	-	not tested
Population density	+	+	+	+	+
Annual rainfall	not tested	not tested	-	-	-
Aptitude of soil for agriculture	not tested	+	+	0	0
Density of protected areas	not tested	-	not tested	not tested	not tested

*Reis and Margulis (1991) use area of municipality in farms

**All studies discount negative results as statistical artefacts

The presence of roads is a strong predictor of deforestation. Previous studies that have focused in particular on the relationship of deforestation to roads have shown that two-thirds of all forest clearing in Amazonia is within 50 km of a major road [42, 5]. When sampling was stratified on deforestation intensity we found that paved roads explained more of the variation in deforestation than did unpaved roads. This agrees with the findings of a study by Laurance and colleagues [43] that showed that paved roads have had much farther-reaching effects on the landscapes they traverse than have unpaved roads. This is probably because many of the roads that are currently paved in Amazonia were major government projects that opened access to previously remote areas ("bringing men without land to land without men" [17]). Principal roads also generally spawn secondary road networks, with settlement and deforestation gradually spreading outward from the initial cuts through the landscape [15]. These principal roads which connect the many smaller networks are probably the most likely to be paved over the long term.

To date, only one empirical study has suggested that paving roads may in fact slow deforestation. The study incorporated over 50 potential predictor variables into a model and examined the relationships of the predictor variables to deforestation [16]. The study found a negative relationship between paved secondary roads and deforestation, which the authors interpret as evidence that paving roads can stimulate agricultural intensification. This interpretation has been used to support the many highway paving projects that are currently underway in the Amazon [44] (see Section 5 for a discussion of current development policy in Amazonia). However, we believe their study suffers from a fundamental weakness: their analyses were based on comparisons of deforestation and other variables at the município (county) level, and because most municípios are located in areas with high population densities and deforestation, their results are unlikely to apply to sparsely populated frontier areas, which are most likely to be impacted by major new highways. Moreover, as interpreted by the authors, the results are generally inconsistent with nearly all other empirical studies of deforestation (table 2).

Overall our study highlights roads as the key component of the deforestation process. Perhaps the most striking finding of our study was the change in the strength of the relationship of deforestation to unpaved roads when sampling was stratified to include a full range of road values. When areas with no unpaved roads and, therefore, no access are explicitly included in the samples of sites analysed, it becomes clear that without road access, there is virtually no colonization and deforestation.

Urban and rural population densities are strongly related to deforestation. Both variables were log transformed, which means that the change in deforestation caused by the first ten people is equivalent to that caused by the 11th to 100th persons, and to the change caused by the 101st to 1000th persons and so on. Our results therefore support the findings of previous studies that the first settlers in a region have a greater effect on the growth of deforested areas than settlers that arrive later on [40]. Again, they highlight the importance of roads to deforestation – the first settlers are unlikely to arrive to a region if it is not accessible by road.

Similarly, there was little additional effect of large-sized cities over small-sized cities on the deforestation of surrounding areas. The highest rates of urbanization in contemporary Amazonia are found in inland settlement frontiers where mining, timber extraction and other resource sectors are the dominant economic activities [45]. The process of farm establishment is ongoing in these areas, and in support of our results Browder and Godfrey [45] have shown that the process of urbanization is associated with higher, not lower, rates of deforestation. Farms owned by urban residents ("absentee owners") are generally more deforested than those owned by rural residents. For example, in a 1990 sample of farms, urban resident farms were 24% more deforested than rural resident farms. This is because urban resident landholders generally employ extensive land uses in order to maintain cleared forest, securing their tenure [45]. By the time cities can be considered 'large' population centres (i.e. more than 100,000 residents), it is likely that most of the immediate surrounding areas have long been farmed, so additional immigrants to large cities would not translate into the same increases in deforestation as would new immigrants to small cities.

We also found that drier, more seasonal forests were more likely to be deforested. Support for a relationship between dry season length and deforestation does not appear to merely reflect the concentration of drier forests along the arc of deforestation. For example, Roraima state, which contains extensive seasonal forests, has experienced high rates of deforestation despite its location in northern Amazonia far from major population centres [41]. Our finding of a more significant relationship between deforestation and annual rainfall when sampling was stratified on rainfall suggests that this sampling regime was better able to capture the significant drop in agricultural land uses in zones with over 2000 mm of rainfall per year [13]. Sombroek [46] hypothesized that high rainfall and short dry seasons may limit agriculture in some regions of the basin because the increased wetness generally means that there is more disease, forest burning is less complete, grains and other crops such as soybeans are more susceptible to rotting, mechanization is more difficult, and rural access roads are difficult to build and maintain.

In contrast to our study, other studies we reviewed that included soil quality as a predictor variable found soil fertility to be positively related to deforestation. There is little detailed information on the distribution of soils in Amazonia, and our lack of a result therefore probably reflects both the coarse scale of our soil data and the large size of our units of analysis (50 km by 50 km grid cells). Soils vary at much finer scales than these, and landholders similarly make decisions to deforest at a scale of a few hectares, and not of a few hundred hectares.

A recent study that used 30 m by 30 m 'pixels' of landscape as its unit of analysis appeared to be much better suited to examining relationships between local biophysical conditions and deforestation [15]. The study used relief as a proxy for soil quality, and found that the presence of relief was a significant deterrent to deforestation. However, the relationship weakened over time, and the authors suggest that as land in their study region became scarce, new colonists were willing to settle in areas with less desirable soils. The change in the strength of the relationship over time is a particularly important finding because it suggests that biophysical variables that may initially discourage deforestation may not deter deforestation as land becomes less available. Other studies have shown that previous clearing is one of the strongest predictors of new deforestation, and that deforestation appears to spread inertially from initially cleared sites [42, 40, 13]. This further suggests that, at a local scale, the influence of biophysical variables on deforestation may weaken over time. The improvement of market, transport and social infrastructure around initial settlements may counteract any negative biophysical aspects of surrounding areas.

In conclusion, at an Amazon-wide scale, we show that proximity to roads is the best predictor of deforestation. Sites that are densely populated, have severe dry seasons and receive less annual rainfall are also more likely to be deforested. Although these relationships were reported by Laurance and colleagues [41], our sensitivity analysis confirms them. Our results are also supported by previous studies. These studies show that when time is taken into account, roads again emerge as the key component of the deforestation process. Population and secondary roads follow [15]. Biophysical variables such as rainfall and soil fertility appear to mediate the extent of deforestation in an area given the presence of these other factors; however the effect of biophysical factors on deforestation may weaken over time as demand for forested land in an area increases.

5. Future of forests in the Brazilian Amazon

Although a goal of Brazilian development policy throughout the last four decades has been to integrate the Amazon into the national economy the scale of the plans for Amazonian development that were unveiled in the country's 2000-2003 federal development plan were unprecedented. "Avança Brasil" (Forward Brazil) included over US$40 billion in planned infrastructure and energy projects for the Legal Amazon region alone [47]. Although a new development plan is set to replace Avança Brasil in 2004, it does not diverge from its predecessor; the majority of investment will be directed to export-oriented activities including projects that will link the soy bean producers of north-central Brazil to Amazon River ports, develop new sources of hydroelectric energy for aluminium processing, and exploit gas reserves in the remote western part of the basin. The new plan includes most of the projects initially proposed under Avança Brasil, as well as some ambitious new projects, such as a road that will connect Roraima state to Georgetown, Guyana [48].

All of the projects proposed under the current pluriannual plan either focus on road improvements or will require road access. The 7,284 km of roads initially listed to be paved under Avança Brazil would almost double the area of forest land that is currently accessible by paved roads, and would come within 50 km of 22 conservation areas and 89 indigenous lands [43, 1]. Furthermore, the roads would result in large-scale fragmentation of pristine areas of the basin [43]. Fragments are more vulnerable to logging, hunting, and fire, and are also more accessible to settlers and land speculators [23, 29, 33, 49]. One research team used spatially explicit models to predict that Avança Brasil, in addition to other federal development projects planned for the Amazon region in 2001, would leave 28% - 42% of the basin deforested or heavily degraded 20 years from now [43].

The wave of new proposed development in Amazonia will take place in the context of already accelerating rates of deforestation (fig. 1). This acceleration can be expected to continue even in the absence of new development projects given the high intrinsic growth rate of the Amazonian population, migration to Amazonia driven by the tremendous number of landless poor throughout Brazil, and the growing dominance of large-scale agricultural activities in old frontiers [50, 51, 24]. Early criticism of Avança Brasil for its environmental short-sightedness and apparent "disconnect[ion] from social and rural development policies that could improve the population's quality of life" triggered a storm of controversy in Brazil [43, 1]. Some of the planned projects have been stalled in order for environmental impact assessments to be carried out while others have yet to be funded. However, many have gone ahead [5].

Government ministries promoting the large-scale infrastructure projects have argued that recent changes in enforcement capabilities, laws and public attitudes will prevent the patterns of deforestation around new roads and highways that developed in the past from arising [52, 53]. However, two recent trends suggest that institutional mechanisms are not yet strong enough to counteract the drivers of deforestation. The first is the dramatic jump in deforestation in 2002, a particularly dry year during which the Brazilian Real was weak and export earnings among soy farmers high. The second is the rush of immigration and deforestation that is already taking place in areas where large projects were planned under Avança Brasil but where they have not yet been initiated (or in some cases even secured funding) [54]. The first migrants to these areas are generally loggers and small farmers drawn by the prospect of future work. Recently released figures for 2003 show that the high rate of deforestation in 2002 was maintained in 2003 [61].

Just as these two trends should be acknowledged by proponents within the government, the finding that biophysical deterrents to deforestation can weaken as demand for land increases [51] should be taken into consideration by those suggesting that the location of new paved roads in wetter areas of the basin will discourage deforestation of adjacent areas [55]. The demand for land in Brazil is very high; in 2003 there were an estimated 25 million landless farmers in Brazil [51].

In appreciation of the development-conservation dilemma faced by the government, some critics of the current development schemes have proposed 'sensitive development' policies that would bring maximum benefits to local populations while causing minimal extra deforestation. For example, it has been suggested that instead of paving federal roads through currently difficult-to-access regions of the basin such as the area traversed by the Cuiabá-Santarém road, road improvements should instead be targeted at secondary roads that connect disjoint rural communities in Amazonia. Carvalho and colleagues [57] argue that this would provide the communities with improved access to markets and social services while causing little superfluous deforestation. Other research has examined the environmental opportunity costs of proceeding with proposed development projects; initial estimates of the economic value of the carbon stocks and biodiversity of the Amazon suggest that investing in their protection might be highly profitable over the long term [2].

Environmental groups had hoped that the presidency of Luiz Inácio Lula da Silva that began in January 2003 would mean a new approach to development of the Amazon

[12]. However, the new, centre-left government is under pressure to fulfil campaign promises to create jobs, feed the urban poor, and boost the national economy; the draft of the 2004-2007 plan reveals the government's decision to rely on large-scale infrastructure projects that aim to increase agricultural production and swell exports [56]. Many of the large-scale projects proposed in the 2000-2003 and 2004-2007 plans are particularly attractive because they are to be funded by international partners and therefore require minimal investments on the part of the government.

In late 2003 those fighting for a re-evaluation of the projects proposed under the 2004-2007 plan were provided with new hope. In their final report, the "Interministerial Working Group on Deforestation", formed by presidential decree in July 2003 to scrutinize the Amazonian deforestation issue, called for the re-evaluation of a number of planned infrastructure projects based on the projects' potential to "open a new front of occupation" and "reproduce the [destructive] model of development which has predominated in Amazonia over the last 20 years." [58]. It remains to be seen how the Lula administration will respond to the report.

Planners and policy makers for the Brazilian Amazon urgently need to consider the potential effects of planned development on Amazonian forests. If minimizing the loss of additional forest areas is indeed to be a federal priority [59], then we make the following recommendations based on the evidence from this study, and the compiled evidence from previous studies.

1. The construction and paving of roads and highways in the Amazon region should be curtailed until environmental impact assessments and economic cost-benefit analyses for local communities have been carried out. All empirical evidence reviewed suggests that paved and unpaved roads are the two factors most strongly

correlated with deforestation, and benefits of road development projects to Amazonian communities are far from certain given the focus of current development plans on providing export-oriented industries from Southern Brazil with quicker access to Amazon River ports. It is key that decisions about roads be made before the roads or highways are built, as once people have access to forests, a Pandora's box of challenges to forest conservation is opened.

2. Efforts by the Brazilian government and external agencies working in conjunction with the government to establish a network of protected areas in the Legal Amazon should focus in particular on forests that experience strong dry seasons. The vulnerability of drier forests to deforestation suggests that deciduous and semi-deciduous forests, woody oligotrophic vegetation (e.g. *campina* and *campinarana*) and ecotonal forests of the cerrado-rainforest interface should be given a priority for conservation. These areas are currently poorly represented in the national system of protected areas [26, 60]. Empirical evidence suggests that classifying areas as reserves discourages deforestation, though not all forms of degradation, even if there is little enforcement of their boundaries [15, 26, 61].
3. Capacity building and engagement of local peoples should be a priority of conservation-oriented activities. In particular, indigenous peoples, whose lands cover 22.5% of the Amazonian biome and overlap with over 70% of Amazonian protected areas, will play a critical role in determining the future of Amazonian forests [26, 27, 61].
4. Further empirical studies are needed to resolve how specific sets of environmental and socio-economic variables combine to determine deforestation at a local scale. However, ample evidence is already available to justify a fundamental change in

current infrastructure-emphasising development programs if additional large-scale deforestation and consequent losses of environmental services are to be avoided.

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Chapter 3

Effects of land use in Eastern Panama on above- and below-ground carbon stocks and tree-species diversity

Kathryn Kirby and Catherine Potvin

1. Introduction

Carbon dioxide is thought to be responsible for 60% of global warming. The majority of anthropogenic carbon emissions – approximately 6.5 Gt C yr^{-1} – are emitted from fossil fuel combustion. However, land cover changes, particularly tropical deforestation, are also important, contributing about 1-2 Gt C yr^{-1}, or 20% of total carbon emissions (IPCC 2000). Recognition of the role of terrestrial ecosystems as either sources or sinks of atmospheric carbon has been accompanied by an active debate over the potential for repairing or enhancing the "sink" potential of these systems, and hence for sequestering atmospheric carbon that would otherwise contribute to global change. Examples might include the reforestation of cleared landscapes, afforestation of unforested landscapes, and the modification of agricultural practices to avoid emissions of carbon stored in soils (Grace 2004). In view of the rate of tree growth and the price of land in the tropics, tropical regions could provide one of the most cost effective opportunities for managing the global carbon cycle (Kauppi and Sedjo 2001). A framework for the provision of incentives for such projects was provided by the Clean

Development Mechanism (CDM) of the Kyoto Protocol. The CDM allows developed countries that agreed to reduce their carbon emissions by 2008 to earn emissions-reduction credits by enabling projects in developing countries that either reduce carbon emissions or create carbon sinks while also contributing to the sustainable development of the host country (UNFCCC 1997).

Although the Kyoto Protocol has yet to be ratified, carbon credits, many of them CDM-eligible, are already being actively traded. In April 2003 the government of Holland had invested $18 million in CDM-eligible, clean energy projects in Panama (ANAM 2003). The Scolel Té project in Mexico, which works with smallholders to establish or enhance carbon sinks in their land, had 400 individual participants representing 30 of the communities in the region in 2003 (Smith and Scherr 2003). The project expected $180,000 in revenue in 2002 for the sale of C services, 60% of which was expected to go directly to participating smallholders (Tipper 2002). In part, the interest in carbon trading to date is speculative: carbon prices will certainly rise if the Kyoto Protocol is ratified. However, many investors are "ethical investors" – corporations or individuals who wish their activities or lifestyles to be carbon-neutral (Grace 2004). Carbon sink projects are particularly attractive to investors interested in protecting multiple ecosystem services; the complexity of vegetation in landscapes managed for carbon generally has concurrent benefits for biodiversity, soil conservation, and local hydrology (de Foresta 1992; Swingland 2003).

Early activity in the carbon market has not been without considerable debate over the technical soundness and social consequences of carbon sink projects. From a policy point of view, issues surrounding project permanence, additionality, and leakage have been major obstacles to the acceptance of the CDM by signatories of Kyoto. However, these issues were largely resolved at the most recent Convention Of Parties meeting (COP 9; Grace 2004). Another technical issue is the uncertainty surrounding the estimate of a project's "baseline" carbon budget, upon which alternative project scenarios (and payments for additional carbon sequestered) are considered. For example, recent studies have emphasized the uncertainty in carbon inventories, in the extrapolation of carbon measurements from sample plots to landscapes, and in the projection of land cover – and hence carbon budgets – through time (Clark et al. 2001; Chave et al. 2004; Pfaff et al. 2004). From a social perspective, the extent to which host countries and local communities benefit from such projects has also received attention (Bass et al. 2000; Tipper 2002; Scherr and Smith 2003; Grace et al. 2004). In particular, these papers argue that carbon projects will only contribute to sustainable development if they are "owned" by their host communities or organisations, and if projects can be maintained without a dependence on foreign experts or "technocrats".

Here we report the technical results of a pilot project that was carried out in cooperation with an indigenous Emberá community in Eastern Panama. The project had three main objectives. First, to develop a method for inventorying carbon that is efficient, allows for easy comparisons among sites and land-use types, and incorporates local ecological knowledge such that it can be carried out by local people with a minimum of training. Second, to determine the average carbon stocks of intact forests, agroforests, and pastures in the collective lands of Ipetí-Emberá, as well as the variation within each. Third, to try and partition this variation into 'environment-determined', 'management-determined' and 'unexplained' fractions, with a view to identifying 'best management' practices for carbon storage among those currently practiced by land owners in the collective lands. A fourth objective of the study, which we explore in parallel to the main three, was to determine how tree diversity is affected by land use, site ecology, and site management.

2.0 Methods

2.1 Study area

The study was carried out in cooperation with the indigenous Emberá community of Ipetí in their tierras colectivas (TC; collective lands) in eastern Panama Province, Panama (N08°58'15.34", W078°31'00.65"). The TC of Ipetí-Emberá encompass 3,198 ha of land in the Bayano watershed, and are framed by the Ipetí river to the east, the Curtí river to the west, and the Interamericana highway to the north (Dalle and Potvin 2004; fig. 1). The rough topography (50-300 m elevation; Instituto Geografico Nacional, 1988) is typical of the foothills of the Serrania de Maje which rises to the south of the TC and forms a barrier to the Pacific Ocean.

The bedrock geology consists of massive conglomerate with basalt boulders and cobbles in a sand matrix of Oligocene age. The highlands to the south of the TC are comprised of older, pre-Tertiary metamorphic and igneous rocks (Stewart 1966). Locally the soil is clay-rich. Mean annual temperature is 26°C, and mean annual rainfall is 2500 mm (Instituto Geografico Nacional 1988). There is a pronounced dry season from December to April (Dalle and Potvin 2004). The area is tropical moist forest according to the Holdridge lifezone system, and the primary forest canopy is approximately 30-40 m tall. Common forest tree species include *Sorocea affinis*, *Gustavia sp.*, *Malayba glaberrima* and *Quararibea asterolepis*. The largest trees in the area are *Anacardium*

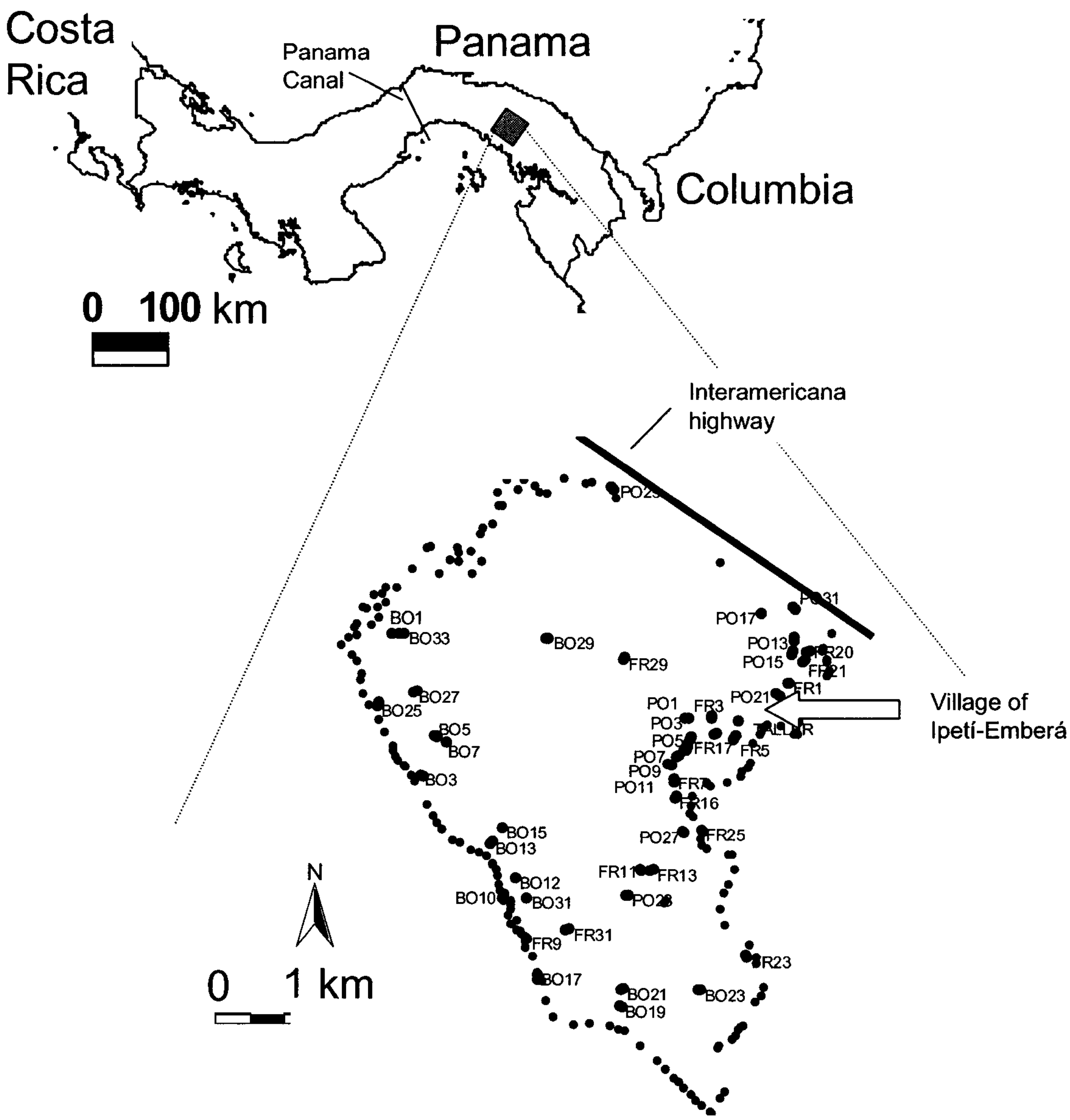

Figure 1. Location of the tierras colectivas (TC) of Ipeti-Embera in Panama, and distribution of sampling sites in the TC. Sampling sites are coded as: BO = forest, FR = agroforest, and PO = pasture. (Or, green = forest, red = agroforest, yellow = pasture). Much of the area in the centre of the map is secondary forest.

excelsum and *Cavanillesia platanifolia*. The palms *Socratea exorrhiza* and *Cryosophila warscewiczii* are the most common understory species.

The TC were designated by the Panamanian government in 1970 for Emberá families whose lands were to be flooded during the construction of the Bayano Dam, which is approximately 40 km to the northwest of the TC (Wali 1993; Dalle and Potvin 2004). In 1970 there were already about 4 Emberá households in the area (pers. comm. Rocali Dumasa). The majority of the relocated families arrived in Ipetí in 1975, and since then the community has grown from approximately 12 households to 71 households and ca. 500 people (P. Tschakert, unpublished data). The TC are subdivided into landholdings ("*parcelas*") that are managed by individual families. Families make independent management decisions regarding their landholdings, but cannot sell the holdings. If a family leaves the community permanently their land is assigned to a new owner by the community's governing body (P. Guainora, pers. comm.). Both birth rates and immigration rates in Ipetí are high, and demand for land in the TC is growing. Most immigrants arrive from Emberá communities closer to the Panama-Columbia border where guerilla-paramilitary violence persists. Newcomers to the TC generally depend on relatives or wage labour for subsistence, as families that had settled in the TC by 1976 continue to hold most of the land (P. Guainora, pers. comm.).

The landscape surrounding the TC is a matrix of forest remnants and cattle pasture that is managed by colonists from central Panama (Dalle and Potvin 2004). In this paper we examine the carbon stocks of three common land-use types in the TC: (1) 'intact' forest (including selectively logged forest), (2) permanent homegarden (agroforest) and (3) pasture. Just under half of the TC is in intact forest. Generally, each extended family in Ipeti manages a permanent homegarden that is comprised of a mix of fruit trees such as mango, orange, mandarin, lime, nance, cacao, coffee, peach palm, coconut, and avocado. In some cases the family may sell part of the fruit harvest for profit. A number of families also grow timber species and medicinal plants in their agroforests. In the last twenty years, a number of landowners in the TC have established pastures for cattle grazing. Landowners who do not have their own cattle may be able to rent their pasture to a cattle rancher from a neighbouring community in return for a monthly or annual rent. Grazing, fumigation, and fire are all used to control unwanted vegetation in pastures.

2.2 Carbon inventory

2.2.1 Local participation

The community *dirigencia* (governing body) and the local development NGO (the Organisacion de la Unidad y Desarollo de Ipetí-Emberá (OUDCIE)) participated actively in the planning of the study, including in the writing of the proposal for project funding. Before the initiation of the inventory work, a general meeting was held at which the goals of the project were outlined to all landowners in the TC, and landowners were asked for permission to inventory their parcelas. The concepts of carbon, its solid and gaseous states, its flux between vegetation, soils and the atmosphere, climate change, and climate change mitigation using terrestrial sinks were explained using locally relevant examples and audience participation. At the same meeting, the community selected six men to make up a carbon inventory team, including two men considered by the community to be experts of local forest trees. The men formed two work teams, each led by one of the two tree-species experts. A week was spent training team members in all aspects of the methodology, with significant time dedicated to concepts such as interpolation between measurement units and conversion between meters, centimeters and millimeters. During the second week of inventory work, the teams inventoried sites independently. These sites were then revisited by the teams and K. Kirby and methods were discussed. Particular attention was focussed on ensuring the sampling sites had been selected using the random methods outlined below, that the paired plots were laid out correctly, and that all trees within the plot boundaries (and no trees outside the plot boundaries) had been measured.

2.2.1 Sampling design

Based on team members' knowledge of current land uses in the TC and a participatory map of land use that was developed in the community in 1998, we identified all landholdings in the TC containing intact forest, agroforest, and/or pasture. For each land-use type the team members and K. K. then randomly selected 16 parcelas using a lottery system, stratifying by sub-watershed to distribute the sampling sites throughout the 3,198 ha of the TC. This scheme was successful in distributing forest sites, however, most agroforests and pastures were concentrated near the village and along the highway, and most of our 16 pasture and 16 agroforest sites therefore fell within this intensively managed zone (fig. 1). At each site the team members and K.K established a pair of circular plots with a radii of 15 m in which all measures of above- and below-ground carbon pools were nested (pair of plots = site). In forest and pasture the centre of the paired plots were separated by 40 m, leaving 10 m between the outside edge of each plot. In agroforests, this method was followed except when the shape of the agroforest prevented it, in which case the two circular plots were placed wherever they fit within the agroforest as long as they did not overlap. In using paired plots for the inventory we were able to capture more heterogeneity than we would have with a single plot of twice the area, and the ease of managing measurements within the smaller plot meant there was little increase in sampling effort.

In each 15 m-radius plot we measured all trees, palms and lianas ≥ 10 cm diameter at breast height (DBH) to the nearest millimeter. At the centre of each plot we established a sub-plot with a 6 m radius in which we measured all trees, palms and lianas 5-10 cm DBH. When buttresses were present, we measured 50 cm above the buttresses (Condit 1998). In the few cases where this was not possible we measured as high as could be reached without a ladder; the number of trees measured in this way was less than 1 per site. We estimated the height of standing trees that had snapped below the crown. Lianas were measured 1.3 m along the stem from the point where they entered the ground (hereafter referred to as liana DBH). From the center of each large plot we laid a 15 m transect in each of the South and East cardinal directions. The diameter of all pieces of downed woody debris (≥ 1 cm diameter) along the 15 m transects was recorded. At the point where each transect intersected the plot perimeter (15 m), we established a 3 m by 3 m quadrat in which we used calipers to measure the basal diameter (BD; diameter at 10 cm above the ground) of all saplings and small palms (< 5 cm DBH, BD ≥ 1 cm) and the diameter of all lianas 1-5 cm DBH. In one randomly selected corner of each 3 by 3 m quadrat we established a 1 m by 1 m quadrat in which we harvested all woody vegetation with BD < 1 cm. Stemless palms were also harvested. Within each 1 m by 1 m quadrat we then established a 50 cm by 50 cm plot in which we harvested all herbaceous plants and collected the leaf litter. Harvested vegetation and litter were dried to constant weight. In each 50 cm by 50 cm quadrat, we also took a vertical soil core at the soil surface (0 cm) and at a depth of 30 cm. The soil cores were 3.0 cm in diameter and 10 cm in height. With 2 soil cores per quadrat (one at each depth), 2 quadrats per plot, and 2 plots per site, we collected eight soil cores per site.

For each tree, palm and liana we measured, the local or Emberá name was recorded by one of the two team members who had been selected by the community as an expert of local plants. Although we originally planned to create voucher specimens, the Chiefs of the Congress of the Comarca Emberá-Wounaan requested that we not. The Congress is concerned that putting any plant specimens from Emberá lands into public herbaria will open the door to biopiracy. Therefore, in order to establish the link between the Emberá nomenclature and scientific taxonomy we used the literature to relate each of the 129 recorded names to a shortlist of scientific (Panama Canal Tree Atlas; Duke and Porter 1970; M. Correa unpublished data). These scientific names were then verified through discussions with botanists at the Smithsonian Tropical Research Institute who are familiar with the flora of Eastern Panama (Appendix 2).

2.2.2 From field measurements to C per ha

Diameter measures of trees, palms, lianas, saplings and woody debris were first converted to measures of above-ground biomass (AGB) per hectare, and then to metric tons (Mg) of carbon per hectare. Before scaling AGB of trees, palms, and lianas in the 15 m- radius and 6 m-radius plots from Mg $plot^{-1}$ to Mg ha^{-1}, the size of each plot was corrected for the steepness of the slope. The method of AGB estimation for each component is described in detail below. For saplings, small lianas, seedlings, herbs, and litter, results from the South and East quadrats were averaged for plot-level analyses, and then scaled to Mg ha^{-1}. Similarly, the AGB of downed woody debris from the south and east transects were averaged for each plot. To convert AGB to C, AGB was multiplied by the % C content of the component in question. C content was assumed to be: 45% for litter, 43% for seedlings, 41% for grass, and 50% for downed woody debris (Hughes et al. 1999). For trees, palms and lianas we assume a C content of 47%. This was the mean C content for palms in a wet forest in Mexico (Hughes et al. 1999), and is intermediate to the mean C content for trees of two neotropical studies (46% in Elias and Potvin (2003) and 48% in Hughes et al. (1999)).

Tree and palm AGB

Two steps in the process of estimating the AGB of trees have recently been emphasized in the literature: the choice of allometric model (Araujo et al. 1999; Keller et al. 2001; Chave et al. 2003; Baker et al. 2004), and the calibration of the allometric model for local conditions (Ketterings et al. 2001; Chave et al. 2003; Baker et al. 2004; Chave et al. 2004). Here we assess the sensitivity of our estimates of tree (including palm) AGB to four alternative allometric models (table 1), and to the assumptions of each model regarding the wood density of our sites, which is one aspect of forest structure that has been shown to vary significantly among tropical forest species and sites (Baker et al, 2004; Muller-Landau et al. 2004; Chave et al. 2003; Chave et al. 2004). Wood density, or wood specific gravity, is the oven-dry weight of wood divided by its wet volume (Fearnside 1997). Models that use only DBH as a predictor variable assume that local wood density is the same as that of the site(s) where the model was originally developed (e.g. Brown 1997; Araujo et al. 1999; Carvalho et al. 1999; Chave et al. 2001; Chambers et al. 2001). Estimates of tree AGB will therefore be inaccurate if wood density differs significantly between the original site(s) and the site to which the model is being applied.

Table 1. Alternative allometric models for trees $\geq$ 5 cm DBH compared in this study

Model			ρ_{av}	C_f	Developed with data	DBH range
(A)	Brown (1997)	$\exp[-2.134 + 2.530\ln(DBH)]*(\rho_i/\rho_{av})*C_f$	0.71[a]	na	pan-tropical, moist forests	>=5 cm[g]
(B)	Chave et al. (2001)	$\exp[-2.00 + 2.42\ln(DBH)]*(\rho_i/\rho_{av})*C_f$	0.61[b]	na	pan-tropical, moist and wet forests	>=10 cm
(C)	Chambers et al. (2001)	$\exp[-0.37 + 0.333\ln(DBH) + 0.933\ln(DBH)^2 - 0.122\ln(D)^3]*(\rho_i/\rho_{av})*C_f$	0.69[c]	1.065[h]	Brazil, moist forest	>=5 cm
(D)	Chave et al. (2004a)	$\exp[-3.742 + 3.450\ln(DBH) - 0.148\ln(DBH)^2]*(\rho_i/\rho_{av})*C_f$	0.6[d]	1.091[h]	pan-tropical, moist and wet forests	>=10 cm
(E)	Chave et al. (2004b)[f]	$\exp[-1.9703 + 2.1166\ln(DBH)]*(\rho_i/\rho_{av})*C_f$	0.54[e]	na	Mexico, wet forest	1-10 cm

DBH = diameter at breast height (cm); ρi = wood specific gravity (g cm-3), i = 1 or 2; ρ1 = species specific value, or 0.54 when wood density of species or species unknown (Chave et al. 2003); ρ2 = 0.499, estimate of average wood density of site (calculated by species, weighted by abundance?); ρav = estimate of the mean wood density of the trees harvested to create the biomass equation; moist tropical forest: 1500-4000 mm rain yr-1; wet tropical forest: >=4000 mm rain yr-1

[a] mean wood density in the data sets of Brown 1997 (not clear whether all data sets, or just those used for this eqn; not clear how weighted) (Ketterings et al. 2001)

[b] neotropical mean (by species, unweighted) (DeWalt and Chave 2004)

[c] mean wood density of plots in central Amazonia (by species, weighted by volume) (Fearnside 1997)

[d] neotropical mean? Chave et al. (2004)

[e] mean wood density of BCI plot (not of plot where equation was developed), used by Chave et al. (2003) to correct this model

[f] Model modified from Hughes et al. (1999)

[h] Published in Chave et al. (2004)

g Brown (1997) recommends using the following equation for trees >=160 cm dbh: AGB = 42.69 - 12.80(DBH) + 1.242(DBH)2, see S 3.1 for discussion

Also relevant to our study are the mean wood densities of the land uses we are comparing; if these differ significantly, a blind application of a DBH-based model might significantly over- or underestimate differences among the carbon stocks of land-use types.

In light of studies showing large variation in wood density among tropical forest sites, the 'average wood densities' of the datasets on which several popular allometric models are based have recently been published (Kettering et al. 2001; Baker et al. 2004; DeWalt and Chave 2004; Chave et al. 2003; Chave et al. 2004). When combined with species-specific wood densities, a simple multiplicative factor of (local wood density)/(average wood density of original dataset) or $(\rho_i)/(\rho_{av})$ can be used to correct each model for local tree densities (Chave et al. 2003; Baker et al. 2004; Chave et al. 2004; DeWalt and Chave 2004). This correction should ideally be applied on a tree-by-tree basis, with ρ_i the specific wood density of tree {i}, and ρ_{av} the volume-weighted mean wood density of the trees used to develop the model (Muller-Landau 2004). Here we linked the species or genus names of the trees we inventoried with species- and genus-specific average wood densities (provided by H. Muller-Landau, J. Chave and colleagues, published and unpublished data)[1]. Although local taxonomies may be less devisive in distinguishing species with similar appearances or uses than are genetic-based taxonomies, wood density is highly conserved within genera (Baker et al. 2004b); our method of linking locally defined morphospecies to scientific species should provide a

[1] Where the shortlist of scientific names included only one species or genus, the wood density for that species or genus was used. Where the shortlist included two species with similar wood densities, the mean of the two species' wood densities was used.

good estimate of the wood density of a given individual[2]. When the species or genus of a tree could not be identified, or when no wood density was available for a species or genus, an average value of 0.54 g cm^{-3} – the average wood density of trees ≥ 10 cm in a plot on Barro Colorado Island, Panama – was assigned to the tree (Muller-Landau 2004). Unfortunately, no estimates of biomass-weighted ρ_{av} for the four models we consider here have been published, and we therefore use the stem- or species-weighted estimates that are available (table 1).

Above ground biomass of dead trees and palms was estimated as for live trees and palms, with values reduced by 10% to account for the loss of leaves, twigs, and small branches (cf. Delaney et al. 1998 in Nascimento and Laurance 2002; table 2). For dead trees that had snapped below the crown, we applied a taper function multiplied by average wood density (Graça et al. 1999 in Nascimento and Laurance 2002; table 2). As with live trees and palms, AGB was calculated using both as-published and calibrated versions of four allometric models, and was added to live tree and palm AGB in each case.

Liana AGB

Liana diameter was first converted to basal area ($BA=\pi*DBH^2/4$) and then to AGB using an allometric model developed in a wet tropical forest in Venezuela (Putz 1983; table 2). We chose the model of Putz (1983) because, like the author, we considered a liana 'individual' to be an independently climbing or self supporting stem. In fact, a single liana may partially fall from the canopy and reroot several times along its

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[2] Baker et al. (2004) found that, among the plots included in the RAINFOR database, "Plots where local names were initially used for the identification of common species have similar levels of resolution [to plots where scientific names were used]"

Table 2. Allometric models used to convert measures of vegetation, litter and debris to AGB

Above ground component	Model	Source
Lianas >= 1 cm	10^(0.12 + 0.91*log(BA))	Putz (1983)
Saplings >= 1 cm BD, < 5 cm DBH - forest + agroforest	exp[3.965 + 2.383ln(BD)]	This study
Saplings >= 1 cm BD, < 5 cm DBH - pasture	exp[3.790 + 2.476ln(BD)]	Potvin, unpublished data
Tree snags >= 5 cm DBH	$\rho_i[\pi(DBH/2)^2*(height)*0.78]$	Nascimento and Laurance (2002)
Dead trees >= 5 cm DBH	90% of total AGB of live tree	Delaney et al. (1998)
Root biomass	24% of AGB of trees, palms, lianas >= 1 cm DBH	Cairns et al. (1997)
Downed woody debris	$\rho_{dwd\_class}[\pi^2\Sigma(d^2)/8L]*C_s$	Brown and Roussopoulos (1974)

DBH = diameter at breast height (cm); BD = basal diameter (diameter at 10 cm above ground level; cm); BA = basal area (cm^2); ρ_i = species specific wood density value (g cm^{-3}) of tree {i}, or 0.54 when wood density of species or species unknown; ρ_{dwd_class} = wood density of downed wood debris class (g cm^{-3}); ρ_{sound_cwd}=0.453; $\rho_{rotting\_cwd}$=0.319; ρfwd=0.453 ; Cs=slope correction factor; L=transect length (cm) length, and some studies are cautious to measure only genetic individuals by locating the ultimate rooting point of each liana (e.g. Gerwing and Farias 2000). Applying a model developed for genetic individuals to our data would result in an overestimate of liana biomass (DeWalt and Chave 2004).

Sapling AGB

Regression analysis (SAS version 8.2) was used to develop an unbiased model relating BD to dry AGB for a sample of 30 saplings (trees <5 cm DBH, ≥ 1 cm BD) from a forest understorey (r^2=0.886, p<0.0001; table 2). The saplings (representing 20 morphospecies) were harvested, separated into trunk, branch, dead wood, twig and leaf components, dried, and weighed (Brown 1997). The model developed was used to estimate the AGB of saplings from forest and agroforest sites. AGB of saplings in pasture sites, which tended to show more horizontal growth, was estimated using a model developed for saplings growing in a pasture in central Panama (r^2=0.856, p<0.0001; C. Potvin, unpublished data; table 2). Young palms encountered in the 3 m by 3 m quadrats were classified into one of two groups. In the first group were individuals with tree-like stems (e.g. most *Bactris spp.* and *Cryosophila warscewiczii* individuals); we estimated their AGB using the allometric models for saplings. The second group included young palms that did not have an obvious stem but had fronds growing from their base (e.g. most individuals of the genera *Attalea* and *Astrocaryum*); individuals in this group were destructively sampled when they fell in the 1 m by 1 m quadrats as part of the 'woody vegetation' component.

Roots

Ninety-two percent of root biomass in tropical deciduous forests is located in the 0-40 cm soil layer (Jobbagy and Jackson 2000). Cairns et al. (1997) reviewed studies in which tree roots had been excavated and their biomass quantified, and found an average root to shoot ratio for tropical forests of 0.24 with a standard deviation of 0.14. This is consistent with the root to shoot ratio of 0.25 that is reported by Jobbagy and Jackson (2000) for tropical deciduous forests. We therefore indirectly estimate the total root biomass for our sites as 24% of the above-ground biomass of trees > 1cm BD. We could not locate any published estimates of root-to-shoot ratios for lianas, and therefore also use this relationship to estimate the biomass of liana roots.

Woody debris

AGB of debris was estimated using the planar-intersect method (Van Wagner 1968, Brown and Roussopoulos 1974; table 2). Debris was classified according to diameter (fine debris 1.0-7.6 cm and coarse debris > 7.6 cm); and coarse debris was further classified as either sound or rotten. Wood densities for the different classes were assigned as follows: 0.319 g cm^{-3} for rotting coarse debris (mean of "partially decomposed" and "fully decomposed" in Clark et al. 2002) and 0.453 g cm^{-3} for sound coarse woody debris and fine debris (Clark et al. 2002). Most pieces of debris lay directly on the forest floor and we therefore consider the error in transect biomass due to tilt of individual pieces to be negligible (Van Wagner 1968). Bias due to transect slope was corrected for each transect.

Soils

Bulk density and pH (using pH electrode; Denver Instrument Company model 300408.1) were determined for every soil core. The organic carbon content of all samples was assessed using a variation of the Loss-on-Ignition method (Ball 1964) as follows: a 1-3 g sample of dried, sieved (< 1 mm) soil was added to an oven-dried, tared crucible and the weight of crucible and soil sample determined. Samples were ignited for 23 hours in a muffle furnace at 375°C, removed from furnace, and immediately weighed. Because water and other inorganic constituents of soil may also be lost upon ignition, inflating estimates of organic carbon (Nelson and Sommers AAS), the cores from one quadrat at each site were also analysed for total organic carbon using a high temperature combustion method (GEOTOP Laboratories, Université de Québec à Montréal). Once eight outlying samples suspected to contain carbonates (pH > 7 or outlying C:N ratio) were removed from the high temperature combustion dataset, regression analysis was used to relate % organic C to weight lost on ignition (LOI), where $\%C = -1.0260 + 0.363(LOI)$ ($r2=0.8327$, $p<0.0001$). This relationship was used to predict % organic C for plots in which no cores had been analysed by high temperature combustion. Soil organic carbon (SOC) per hectare per layer (0-10 cm or 30-40 cm) was determined according to the following formula: [%C]*[bulk density]*[depth of layer], with bulk density measured in g cm^{-3} and depth of layer in cm. As a rough estimate of SOC in the 10-30 cm layer we took the mean SOC value for the 0-10 and 30-40 cm layers, and applied it to each of the 10-20 and 20-30 cm layers. We found that this method produced summed values for the 0-20 and 20-40 cm layers in forest sites that were consistent with previous accounts of the relative distribution of SOC with depth in tropical deciduous forests (Jobaggy and

Jackson 2000), with the 20-40 cm layer in our forest sites making up 41.1% of the 0-40 cm layer.

2.3 Site ecological characteristics and management histories

At the centre of each plot we recorded latitude and longitude using a Global Positioning System (Garmin 120). Ordinal measures of soil texture, topographical position (ridge, slope, valley) and slope steepness were also recorded for each plot, always by the same team member to ensure consistency. We carried out semi-structured interviews with landowners regarding the management history of their site. Landowners of forest sites were asked whether they harvested fuelwood or timber from their land. Agroforest and pasture owners were asked when the primary forest on their land had originally been cleared, when the current land use had been established, and what the duration of all intervening land uses had been. Current management practices, including thinning, fumigation, and burning, were also discussed. Agroforest owners were asked whether they sold any of the produce from their agroforests, and whether they grew (and had ever harvested) timber species in their agroforest. Pasture owners were asked if their pasture was grazed, and if so with what regularity and by how many cattle on average. All landowners were asked what they planned to do with their land in the future.

2.4 Analysis

The estimates of forest tree AGB at the plot-level by models A-D (table 1) were compared using paired t-tests with a bonferroni correction ($\alpha = 0.004$). Above- and below- ground carbon pools were contrasted among forest, agroforest, and pasture sites using nested analyses of variance (ANOVAs). Data for woody debris, trees 1-5 cm DBH and trees 5-10 cm DBH, could not be transformed to meet assumptions for parametric tests, and were therefore assessed by means of a Kruskal-Wallis test. Nested ANOVAs allowed variation in the response data to be partitioned between 'land use' and 'site' effects. This was not possible with the Kruskal-wallis tests as these required that results be averaged by site. Where the tests indicated significant differences among land-use types, means were contrasted with post hoc Tukey HSD tests ($\alpha = 0.05$) or, for non-parametric data, with two-way post hoc Mann-U Whitney tests. Variation in C pools among sites in each land-use type was contrasted with a Bartlett's test for homogeneity of variances. Analyses were carried out with the UNIVARIATE, NESTED, NPAR1WAY, and GLM procedures in SAS version 8.2. A post-hoc sample size analysis was used to determine the number of sites necessary for a given level of precision [$n = (s/x)^2(t^2_\alpha/r^2)*100^2$] or for a given absolute error [$n = (t_\alpha s/d)^2$], where n is the sample size needed to estimate the mean, t_α is the student t-value for n-1 degrees of freedom for the 1-α level of confidence, s is the standard deviation of the variable, r is the desired relative error (width of confidence interval as a percentage) and d is the desired absolute error (Krebs 1999).

A matrix of management variables by site was developed based on interviews with landowners. Nominal yes/no responses were coded as 1/0, while quantitative data (such as site age, or times thinned per year) were maintained in their original form. Site ecological characteristics were based on averages of each characteristic for the two plots at a site. We looked for environmental and management correlates of above-ground biomass and % C and bulk density in the 0-10 and 30-40 cm layers. This was done through forward selection of all environmental and management variables (table 3) on the response variables after checking scatterplots to ensure that assumptions of normality and homogeneity of variances were met in each case.

Table 3. Management and environment variables included as explanatory variables in multiple regression analyses.

Variable	Description	Data type	Forest	Agroforest	Pasture
Envrionmental variables					
% N	% N in soil surface layer	Quantitative	x	x	x
pH	pH of soil surface layer	Quantitative	x	x	x
Redness	Redness calculated from munsell colour value, hue, and chroma	Quantitative	x	x	x
Clayiness	Measure of percent clay in soil	Ordinal	x	x	x
Drainage	Site located on ridge, slope, valley	Ordinal	x	x	x
Steepness	Steepness of slope; 1 = < 10% slope; 2 = 10-15% slope; 3 = 15-30% slope; 4 = 30-55% slope; 5 = > 55% slope	Ordinal	x	x	x
Treefall gap	Presence/absence of gap	Nominal	x		
Stream	Presence/absence of permanent or seasonal stream	Nominal	x	x	
Rocks	Presence/absence of rocks in soil	Nominal	x	x	
Management variables					
Distance to village	Distance of site from village	Quantitative	x		
Distance to border	Distance of site from closest border of TC	Quantitative	x		
Timber harvested	Timber harvested from site	Nominal	x	x	
Coffee	Coffee sold from agroforest	Nominal		x	
Orange	Oranges sold from agroforest	Nominal		x	
Avocado	Avocado sold from agroforest	Nominal		x	
No fruit sold	All fruit consumed within household	Nominal		x	
Fine wood	Fine wood grown in agroforest	Nominal		x	
Burned in 2003	Burned in fires of March 2003	Nominal		x	
Frequency of cleaning	Times understory cleaned with machete per year	Quantitative		x	
Fallow prior to current land use	Fallow prior to current land use	Nominal		x	x
Years since primary forest cleared	Years since primary forest cleared	Quantitative		x	x
Years since current land use established	Years since current land use established	Quantitative		x	x
Grazed	Site grazed by cattle	Nominal			x
Fumigated	Site fumigated	Nominal			x
Burned annually	Site burned annally	Nominal			x
Rented	Site rented to neighbouring cattle ranchers	Nominal			x

We also used Pearson correlation analyses to look at relationships among stem density, AGB, and morphospecies richness among plots. Because not all individuals in our plots were identified with a common name, we estimate number of morphospecies per plot by taking the average of the "maximum" number of morphospecies, (assumes that all unidentified woody stems are different morphospecies) and the "minimum" number of morphospecies (assumes that all trees, palms, or lianas in a plot that were not assigned a common name belonged to the same unknown species). On average, in all land-use types, less than 2% of individuals ≥ 10 cm DBH in a plot were unidentified; and in only 3 of the 96 sample plots were more than 6% of individuals unidentified.

Linear regression analysis was used to relate soil properties to time since forest clearance and time since establishment of current land use. Linear regression analysis was also used to relate AGB of forest sites to the distance of the site from the village (distances were measured "as the crow flies" in Arcview version 3.2), and logistic regression was used to relate the history of timber harvesting of different sites (yes/no) to distance of the site from the village. Throughout the text, mean values are presented ± 1 standard error.

3. Results

3.1 Inventory methods

3.1.1 Sampling design

In addition to determining the carbon stocks of the three land-use types we inventoried, a primary objective of our pilot study was to assess our methodology in terms of its accuracy, accessibility to local people, and its incorporation of local ecological knowledge. A post-hoc sample size analysis of our results for tree AGB (results presented in S 3.1.2) suggests that, using our sampling design, we would need 91 forest sites to know the mean AGB of trees $\geq$ 10 cm DBH with 20% error (95% confidence interval +/- 10% of the mean). For the same level of precision in the estimate of tree AGB of agroforests and pastures, we would need 55 agroforest sites and 1971 pasture sites.

The inventory teams were confident in the sampling methodology and able to work independently by the third week. In terms of its ability to incorporate local ecological knowledge, 2,374 of the 2,468 trees $\geq$ 5 cm DBH that we inventoried were identified with a local name. The local names corresponded to 129 morphospecies, 76 of which were linked to a scientific species, and 88 to a genus. This data allowed us to assign wood density values to all but 475 stems representing 16.7% of the total basal area of trees and palms $\geq$ 5 cm DBH across all plots. As described in the methods, the average wood density value for species on BCI (0.54 g cm-3) was used for these remaining individuals.

3.1.2 Tree and palm AGB

Average AGB per hectare of trees and palms $\geq$ 10 cm DBH varied by up to 37% within a land-use type when calculated with the alternative allometric models (models A-E, table 1; results, table 4). Model (A), which has an exponential form, produced very high estimates of AGB for trees with large diameters, and therefore much higher estimates of AGB per hectare of forest than the other three equations (fig. 2a). For agroforests, which had few large trees, (A) produced lower estimates of AGB than (C); this is the pattern we expected to see based on previous comparisons of the same

Table 4. AGB (Mg ha^{-1}) of trees $\geq$ 10 cm DBH $\pm$ 1 SE as predicted by four allometric models

Model	As published	Corrected with ρ_i/ρ_{av}[a]
Forest		
(A)	536.7 ± 68.8	394.9 ± 54.2
(B)	381.1 ± 45.6	327.4 ± 42.1
(C)	393.2 ± 27.9	328.7 ± 26.2
(D)	338.2 ± 34.8	325.9 ± 36.5
Agroforest		
(A)	151.0 ± 11.9	117.4 ± 10.7
(B)	118.8 ± 8.8	107.5 ± 9.2
(C)	189.5 ± 13.6	161.7 ± 13.7
(D)	121.4 ± 9.0	121.9 ± 10.6
Pasture		
(A)	5.0 ± 2.5	3.6 ± 1.9
(B)	4.0 ± 2.0	3.3 ± 1.7
(C)	6.3 ± 3.2	5.0 ± 2.6
(D)	4.0 ± 2.0	3.7 ± 1.9

[a] and Cf if available (see table 1)

(a) (b)

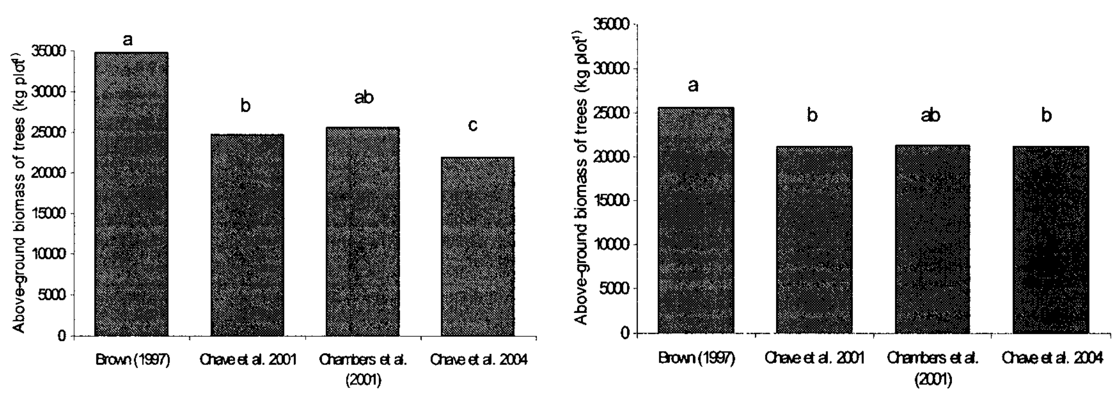


Figure 2. Estimates of mean AGB of forest plots using four alternative allometric models (Brown 1997; Chave et al. 2001; Chambers et al. 2001; Chave et al. 2004). Models are either applied as published (fig. 2a) or corrected for wood density (fig. 2b). Small letters show mean differences among models' estimates that are significantly different than zero in paired t tests (α value corrected for multiple comparisons such that α=0.004).

allometric models (Chave et al. 2003). Even when we corrected (A) by using the equation recommended in Brown (1997) for trees ≥ 160 cm (footnote table 1), the model's estimate of forest AGB was still at least 19% higher than the next highest estimate. Paired t tests of tree AGB estimates at the plot level by the four models revealed that the estimates of model (A) were significantly different than those of (B) and (D), but not significantly different then those of (C) [...but only when a bonferroni correction was applied, lowering the α-value to 0.004] (fig. 2a). The estimates of forest tree AGB of models (B) and (C) were significantly different then those of (D) but not from each other. Once corrected for wood density, the estimates of AGB for forest sites of these three models varied by less than 1%, and were not significantly different from one another (fig. 2b). However, the estimates of (B) and (D) still differed from those of (A) (fig. 2b). Across the four models, the correction for wood density reduced estimates of AGB for trees ≥ 10 cm DBH in all land-use types by an average of 11%. While equation (B) changed substantially when corrected for local wood density, equation (D), which had the same mean site wood density as equation (B), changed very little. The low sensitivity of equation (D) to differences in wood densities among sites may reflect its conservative estimates of large tree AGB (fig. 3). When the estimates of tree AGB of all land-use types are considered, the estimates of equation (D) are generally intermediate to those of (B) and (C). Hereafter, any results for AGB of trees ≥ 10 cm DBH are calculated using equation (D) corrected for local wood density. For trees ≥ 5 cm DBH, we estimate AGB using the wood-density corrected version of equation (A), which generally produced estimates intermediate to those of (C) and (E) (table 5). Both equations (A) and (D) are pan-tropical models.

Table 5. AGB (Mg ha^{-1}) of trees 5-10 cm DBH ± 1 SE as predicted by three allometric models

Model	As published	Corrected with ρ_{tree}/ρ_{av} [a]
Forest		
(A)	9.82 ± 1.25	7.48 ± 0.96
(C)	11.28 ± 1.45	9.40 ± 1.22
(E)	4.96 ± 0.61	4.97 ± 0.62
Agroforest		
(A)	5.95 ± 2.06	4.64 ± 1.44
(C)	6.83 ± 2.37	5.84 ± 1.82
(E)	3.00 ± 1.02	3.07 ± 0.94
Pasture		
(A)	0.31 ± 0.23	0.17 ± 0.13
(C)	0.36 ± 0.27	0.21 ± 0.16
(E)	0.15 ± 0.11	0.11 ± 0.08

[a] and Cf if available (see table 1)

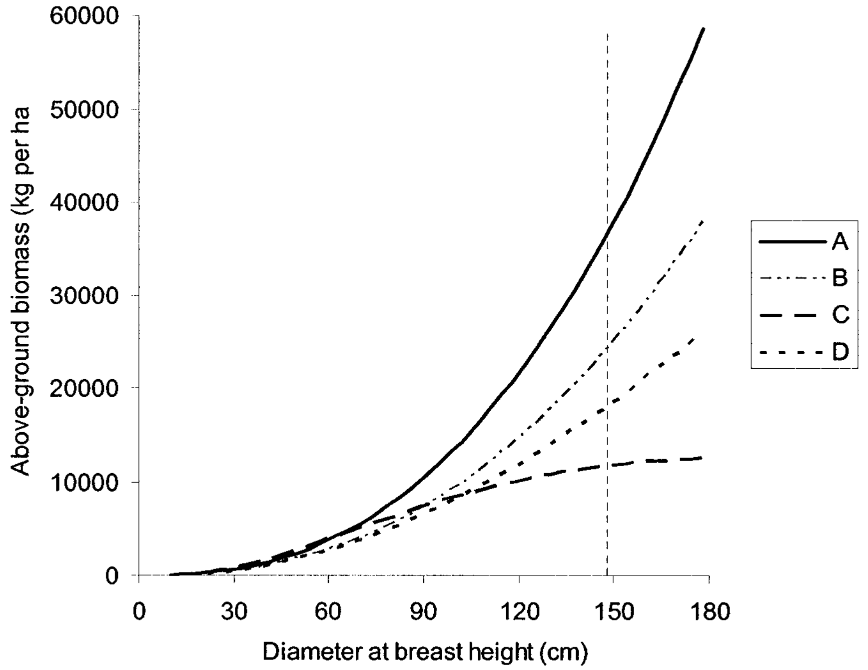


Figure 3. Plot of the AGB estimate of four allometric models for a given dbh. The vertical line indicates the maximum dbh of trees used to develop models (A), (B), and (D); AGBs for trees with larger diameters are extrapolated. The maximum dbh of trees used to develop model (C) was even smaller. The maximum dbh of trees in our study was 178 cm.

3.2 Variation in carbon stocks and diversity among land-use types

The ANOVAs of AGB (table 6) indicate that intact forest had significantly more AGB, and therefore above-ground carbon, than agroforests, which in turn had significantly more AGB than pastures. Soil organic carbon (SOC) did not differ significantly among land-use types (table 7). However, bulk density and % C, which are the variables used to calculate SOC, displayed opposite trends in the surface soil layers of the three land-use types: while pasture soils were more compacted than agroforest soils, which were more compacted than forest soils ($P=0.0224$), pasture soils were 14% lower in C than agroforest soils, which were 5% lower in C than forest soils ($P=0.3251$; table 7). While the variation among sites within a land-use type was not significant for any of the above-ground components that we measured, the variation in bulk density and % C at the surface layer, and for bulk density at 30-40 cm depth differed significantly among sites within a land-use type (table 7). This suggests that initial differences in below-ground variables among sites may obscure significant changes in the variables following land-use change.

When the mean values for all above-ground and below ground pools are summed by land use, the total carbon stocks are 255 Mg ha^{-1} for forests, 127 Mg ha^{-1} for agroforests, and 45 Mg ha^{-1} for pastures (fig. 4). These values include estimates of root biomass as well as estimates of SOC to 40 cm depth (see S 2.2).

The variation in each C pool was significantly different among land-use types for all above ground components ($p<0.001$ in all cases; table 8). In land-use types where a

Table 6. AGB of vegetation, litter and debris among land use types (± 1 SE). Post hoc means comparisons show significant differences among land use types (F=forest; A=agroforest; P=pasture).

	Total AGB	Trees >= 10 cm DBH	Trees 5-10 cm DBH	Saplings	Lianas >= 1 cm DBH	Seedlings	Herbs	Litter	Woody debris
Forest	362.2 ± 36.5	325.9 ± 36.5	9.1 ± 1.2	4.5 ± 0.5	8.8 ± 2.1	0.34 ± 0.04	0.26 ± 0.08	5.25 ± 0.48	8.09 ± 1.87
Agroforest	142.7 ± 10.7	121.9 ± 10.6	5.7 ± 1.8	1.2 ± 0.3	0.4 ± 0.2	0.32 ± 0.05	0.35 ± 0.07	5.84 ± 0.57	7.06 ± 1.58
Pasture	8.4 ± 2.2	3.7 ± 1.9	0.2 ± 0.2	0.4 ± 0.2	0 ± 0	0.37 ± 0.11	2.71 ± 0.39	0.53 ± 0.14	0.41 ± 0.31
Post-hoc	F>A>P	F>A>P	F>A>P*	F>A>P*	F>A>P*	NS	P>A,F	A,F>P	F>A>P*
LU F stat	348.45	481.59				0.03	68.58	107.10	
LU (P value)	(<0.0001)	(<0.0001)				(0.9706)	(<0.0001)	(<0.0001)	
Site F stat	1.39	0.72				2.27	2.76	1.44	
Site (P value)	-0.132	-0.8643				(0.003)	(0.0003)	(0.1087)	
Percent variance explained									
LU	92.66	91.54				0.00	75.61	79.64	
SITE	1.20	0.00				61.25	11.41	3.66	
Unexplained	6.14	8.46				38.75	12.98	16.70	
*Non-parametric means comparison (Mann-U Whitney test)									

Table 7. Results for bulk density, % C, soil organic carbon (SOC), AND % N at depths of 0-10 cm and 30-40 cm among land use types (± 1 SE). Post hoc means comparisons show significant differences among land use types (F=forest; A=agroforest; P=pasture).

	Bulk density 0cm	% C 0 cm	SOC 0 cm	Bulk density 30 cm	% C 30 cm	SOC 30 cm	% N 0 cm	% N 30 cm
Forest	0.49 ± 0.01	3.19 ± 0.20	15.30 ± 1.09	0.63 ± 0.03	1.18 ± 0.11	7.23 ± 0.64	0.31 ± 0.02	0.11 ± 0.01
Agroforest	0.54 ± 0.03	3.04 ± 0.35	15.14 ± 0.84	0.59 ± 0.03	1.32 ± 0.14	7.37 ± 0.63	0.30 ± 0.03	0.12 ± 0.01
Pasture	0.60 ± 0.04	2.62 ± 0.26	14.62 ± 0.80	0.54 ± 0.03	1.13 ± 0.11	5.84 ± 0.40	0.25 ± 0.02	0.10 ± 0.01
Post-hoc	P>A>F	NS	NS	NS	NS	NS	NS	NS
LU F stat	4.14	1.15	0.15	1.92	0.68	2.32	1.76	0.83
LU (P value)	(0.0224)	(0.3251)	(0.8621)	(0.1588)	(0.5119)	(0.1095)	(0.1842)	(0.4425)
Site F stat	7.11	2.88	1.39	5.59	1.56	0.94	2.72	1.17
Site (P value)	(<0.0001)	(0.0002)	(0.1307)	(<0.0001)	(0.0668)	(0.5881)	(0.0004)	(0.2953)
Percent variance explained								
LU	14.67	0.70	0	4.64	0.00	3.73	3.34	0.00
SITE	64.28	48.05	16.38	66.40	21.77	0	44.68	7.87
Unexplained	21.05	51.25	83.62	28.96	78.23	96.27	51.98	92.13

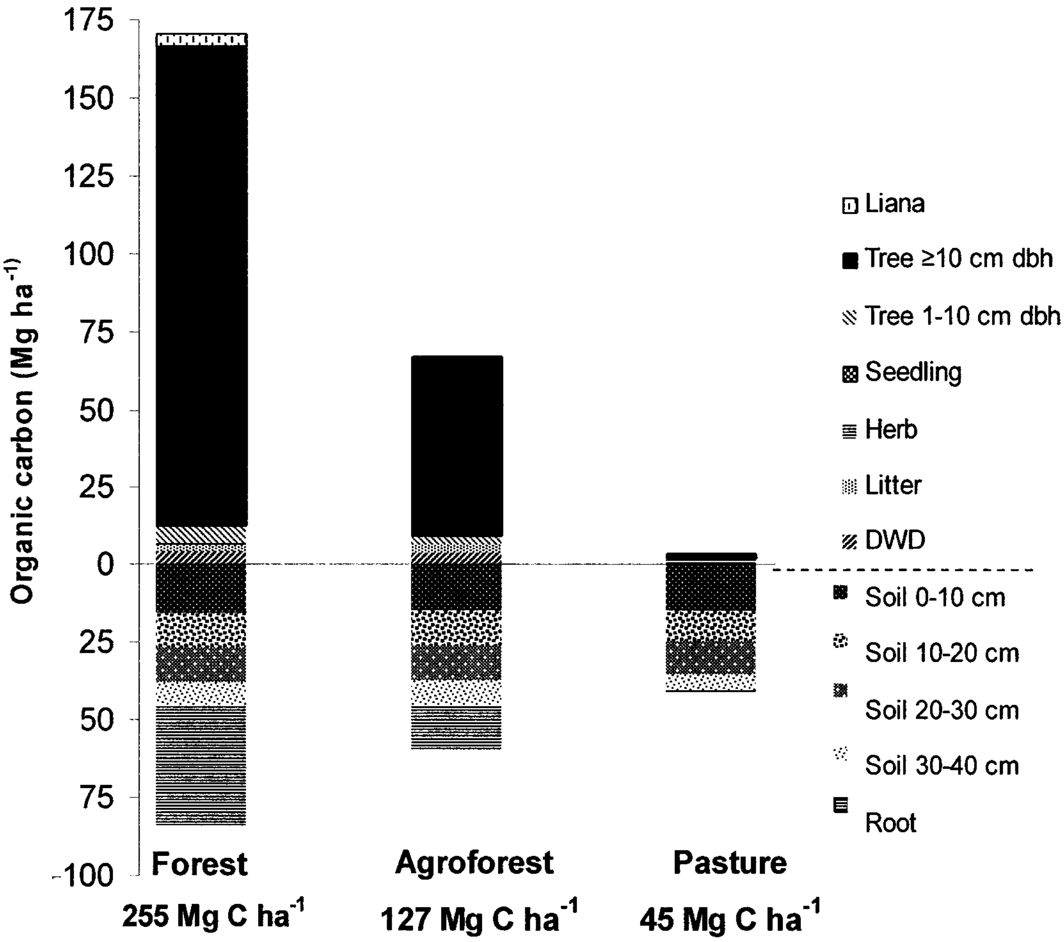


Figure 4. Carbon pools of forests, agroforests and pastures

component was "rare", such as downed woody debris in pasture sites, it generally had a much higher coefficient of variation.

Land use accounted for 91.5% of the variation in trees $\geq$ 10 cm DBH ($P < 0.0001$; table 6), and trees $\geq$ 10 cm DBH were the component that accounted for most of the difference in total C stocks among land-use types (fig. 4). The mean wood densities of the land-use types that we compared were not significantly different (table 9). Breaking trees, including palms, into size-class bins reveals that the relatively low C stock of pastures is a result of the near absence of stems $\geq$ 10 cm

Table 8. Coefficient of variation for carbon stocks in forest, agroforest and pasture sites. Bartlett's test for homogeneity of variance shown for each carbon component.

	Trees >= 10 cm DBH	Trees 5-10 cm DBH	Saplings	Lianas >= 1 cm DBH	Seedlings	Herbs	Litter	Woody debris	Bulk density 0cm	% C 0 cm	SOC 0 cm	Bulk density 30 cm	% C 30 cm	SOC 30 cm
Coefficient of variation														
Forest	44.81	51.77	45.92	95.35	47.35	125.30	36.45	92.66	10.88	24.95	28.56	16.84	37.14	35.13
Agroforest	34.68	124.54	87.42	180.80	57.92	78.29	39.05	89.60	20.99	45.63	22.32	21.80	42.73	34.03
Pasture	208.37	304.22	202.23	-	121.60	58.27	101.12	305.47	24.54	39.76	21.98	24.03	39.22	24.59
Bartlett's test for homogeneity of variances														
Chi-Square	76.999	43.894	14.894	52.581	18.886	52.715	23.771	33.324	13.163	4.364	1.656	0.751	1.269	5.271
Pr > Chisq	<.0001	<.0001	0.0006	<.0001	<.0001	<.0001	<.0001	<.0001	0.0014	0.1128	0.437	0.6869	0.5302	0.0717

Table 9. Average wood density per land use type. Weighted by basal area, AGB (calculated using Chave et al. 2004), and number of stems (± 1 SE).

	Sites[a]	Average wood density (weighted by basal area)	Average wood density (weighted by volume)	Average wood density (weighted by number of stems)
Forest	16	0.534 ± 0.0154	0.535 ± 0.016	0.521 ± 0.015
Agroforest	16	0.548 ± 0.010	0.546 ± 0.010	0.552 ± 0.007
Pasture	9	0.511 ± 0.032	0.509 ± 0.033	0.517 ± 0.031

[a] Seven of the pasture sites did not contain trees

DBH in pasture sites, whereas the difference between agroforests and forests is explained by the near absence of stems ≥ 50 cm DBH in agroforests (fig. 5). When trees and palms are considered separately, the number of trees ≥ 10 cm DBH per hectare did not differ significantly among forests (375 ± 14) and agroforests (392 ± 36), though basal area of trees in a hectare of forest (36.2 m^2 ± 3.4) was more than double that of trees in a hectare of agroforest (15.7 m^2 ± 1.2) (table 10). In contrast, there were significantly more palm stems ≥ 10 cm DBH per ha of forest (140 ± 25) than agroforest (36 ± 10), however the basal area of palms in forests and agroforests did not differ significantly (table 10). This is a reflection of the dominant palms in each land-use type; in forests the small-diameter, understorey palms *Socratea exorrhiza* and *Cryosophila warscewiczi* are most numerous, whereas in agroforests the canopy-reaching *Cocos nucifera* and *Bactris gasipaes* dominate. We encountered no lianas ≥ 10 cm DBH in any of the agroforest or pasture plots.

The number of morphospecies also differed significantly among plots, with an average of 244 ± 9 per ha of forest, 123 ± 7 per ha of agroforest, and 10 ± 3 per ha of pasture (table 10). Land use explained 89.9% of the variation in morphospecies richness, the remainder of the variation was unexplained ($P < 0.0001$; table 10). AGB of trees (including palms) was uncorrelated to either number of stems or number of morphospecies in forest or agroforest sites. However, the relationship was significant in pastures (AGB:stems $r=0.58$, $p=0.019$; AGB:morphospecies $r=0.70$, $p=0.0025$). Over all plots, 93 morphospecies were identified in forests, 59 in agroforests, and 19 in pastures. Of the 59 morphospecies encountered in agroforests, 29 of the morphospecies were never found in forest plots, and 11 of the 29 were species not native to the Neotropics. In pastures, 7 of the 19 morphospecies were never found in forests, and 2 of the 7 were exotics (fig. 6).

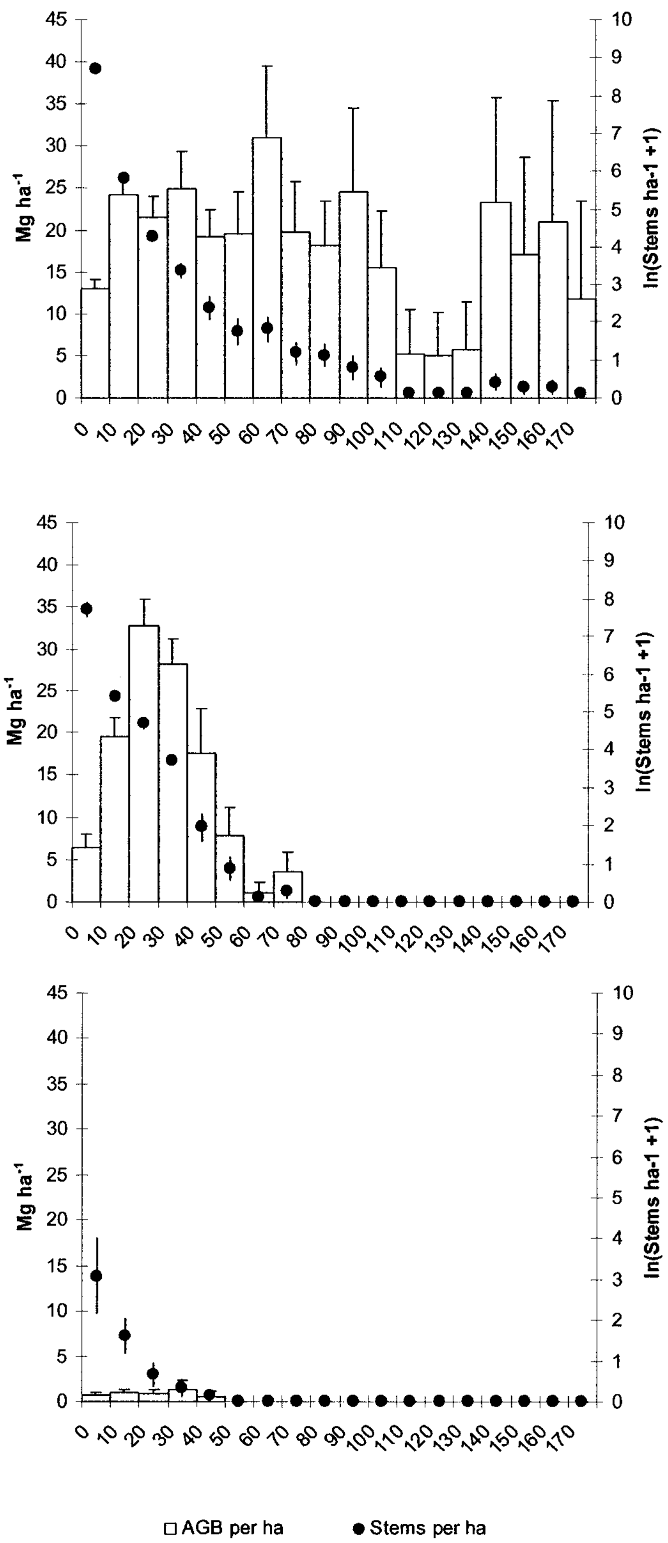
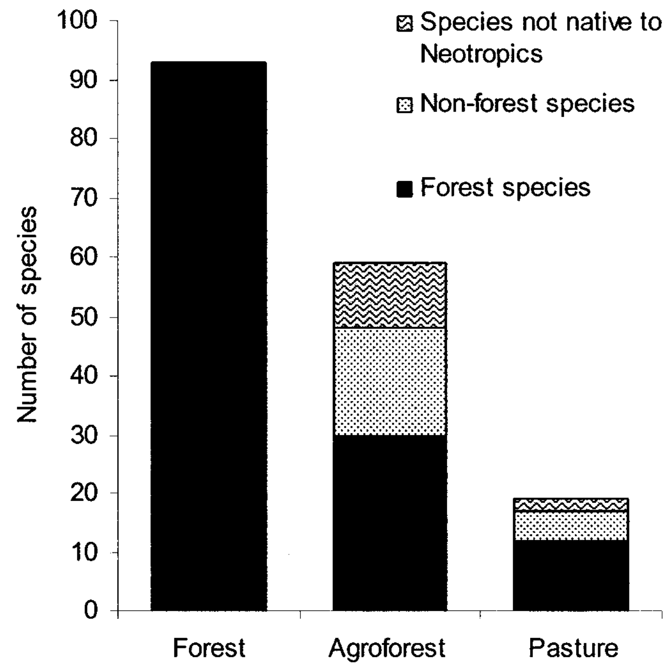


Figure 5(a-c). Distribution of biomass and stem numbers among 10 cm diameter classes in (a) forests, (b) agroforests, and (c) pastures.

Table 10. Number of woody stems and morphospecies ≥ 10 cm DBH per hectare (± 1 SE)

	Trees (no palms) >= 10 cm dbh	Palms >=10 cm dbh	Lianas >= 10 cm dbh	Basal area (m^2) of trees >= 10 cm dbh	Basal area (m^2) of palms >= 10 cm dbh	Basal area (cm^2) of lianas >= 10 cm dbh	Number of morphospecies (woody stems >= 10 cm dbh)
Forest	375 ± 14	140 ± 25	15 ± 5	36.2 ± 3.4	2.6 ± 0.6	0.19 ± 0.07	244 ± 9
Agroforest	392 ± 36	36 ± 10	0 ± 0	15.7 ± 1.2	1.6 ± 0.4	0 ± 0	123 ± 7
Pasture	19 ± 6	1 ± 1	0 ± 0	0.57 ± 0.23	0.034 ± 0.029	0 ± 0	10 ± 3
Post hoc	A,F>P	F>A,P	F>A,P	F>A>P	F,A>P	F>A,P	F>A>P
LU F stat	84.79	21.39		73.15	8.05		288.35
LU (P value)	(<.0001)	(<.0001)		(<.0001)	(0.001)		(<.0001)
Site F stat	3.56	1.89		0.58	1.77		0.99
Site (P value)	(<.0001)	(0.0159)		(0.9651)	(0.0265)		(0.506)
Percent variance explained							
LU	80.34	45.45		56.77	21.99		89.93
Site	11.03	16.78		0.00	21.72		0.00
Unexplained	8.63	37.77		43.23	56.29		10.07



	Total species	Forest species	Non-forest species	Species alien to Neotropics
Forest	93	93	0	0
Agroforest	59	30	18	11
Pasture	19	12	5	2

Figure 6. Total number of species identified with a common name from all plots.

3.3 Variation in carbon stocks and diversity within land-use types

3.3.1 Site characteristics

Forests tended to be on steeper slopes. Neither pH nor soil colour differed significantly among land-use types (pH: P=0.2501; Munsell redness: P=0.5484). All forest owners reported having cleared areas of primary forest in their parcela for annual crops. Fifty-three percent of forest owners reported harvesting timber from their forest parcelas for construction or resale, and 20% reported that wood from their parcelas had been harvested illegally by ranchers or farmers living outside the TC. The average time since primary forest had been cleared for agroforest and pasture was 26.2 ± 6.2 and 23.3 ± 6.4 years respectively. Typically, agroforests were planted after two years of annual cropping. Fifty-six percent of agroforest owners reported selling part of their fruit harvest for profit, and in all cases but one the fruits sold were oranges, coffee, or avocados. The average age of pastures was 12.5 ± 6.1 years. During the period between primary forest clearing and pasture establishment, most pasture owners reported having planted rice and/or corn for two years, and then a root crop such as ñame (*Dioscorea sp.*). During ñame cultivation the forest was allowed to regenerate, and after two years the land was left fallow. According to the typical agricultural cycle in the TC, the land would have been left fallow until soils were deemed sufficiently recovered, or need was sufficiently great, for a new round of agriculture. In the case of the pasture owners interviewed, the owners had decided to truncate the crop rotation cycle and establish pasture instead. When the study was conducted, 43% of pasture owners owned cattle, and another 31% were renting their pastures to cattle ranchers from outside the community.

3.3.2 Role of environment and management in determining C stocks

None of the measured environmental or management variables explained a significant portion of the variation in above-ground biomass in the forests, agroforests or pastures. AGB of agroforests tended to increase with agroforest age (r^2=0.46473; p=0.0941; fig. 7), however trees are added to agroforests through time, and the age of trees in an agroforest is therefore highly variable. The higher variation in AGB of older agroforests than in AGB of more recently established agroforests reflects this practice (fig. 7).

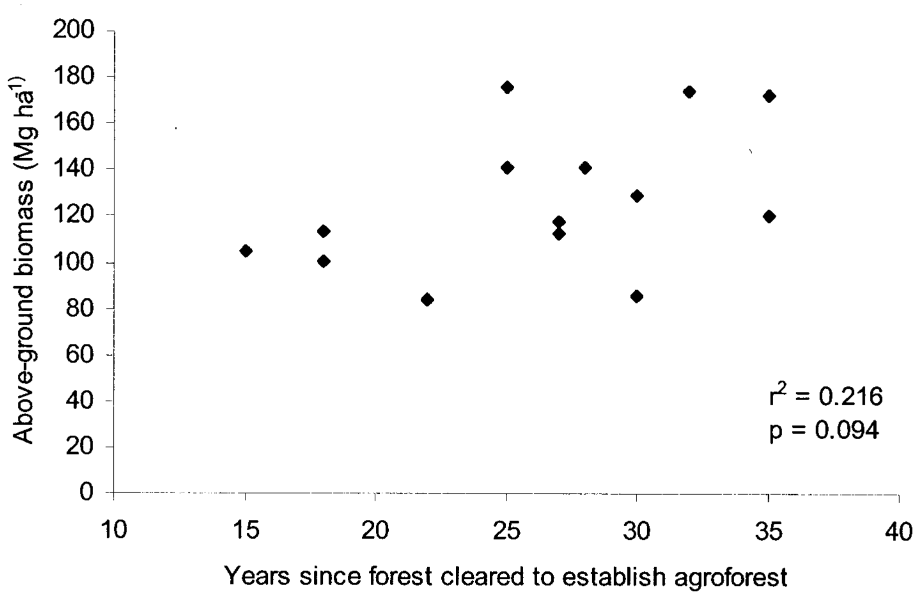


Figure 7. Total above-ground biomass of agroforests with time since primary forest cleared to establish the agroforest.

Percent C in the 0-10 cm layer of soil was positively correlated to % N in that layer in all three land-use types (Forest: r^2=0.5278, p=0.0014; Pasture: r^2=0.8426, p< 0.0001). In Agroforests, % N explained the majority of variation in % C in the 0-10 cm layer (r2=0.913, p<0.0001), however an additional 5% of the variation was also explained by the frequency of thinning with machete (p=0.0175). % C in the 30-40 cm layer in forests and pastures was not explained by any of the measured variables. There was a weak trend of decreasing % C with pasture age, but the trend was not significant. In agroforests % C at 30-40 cm was negatively correlated to pH, positively correlated to % N in the 0-10 cm layer and negatively correlated to market-oriented management (r^2=0.760 p(pH)=0.0118, p(N)=0.0027, p(no_sell)=0.0006). More intensive management of agroforests (i.e. more frequent thinning or market-oriented management) therefore seemed to be associated with higher % C in agroforests at both the 0-10 cm and 30-40 cm depths.

Bulk density was negatively related to % N in agroforests in the 0-10 cm layer (r^2=0.4891, p=0.0026) and in pastures in both the in the 0-10 cm and 30-40 cm layers (0-10: r^2=0.3700, p=0.0158; 30-40: r^2=0.2991, p=0.0284). Bulk density at the 0-10 cm layer in forests was positively related to pH (r^2=0.3609, p=0.0139). The opposite relationships of % C and bulk density to %N in pastures and agroforests again emphasises that considering only SOC may obscure trends in soil C, including relationships of soil C to management or environmental variables. In pastures, steepness of slope explained an additional 23% of the variation in bulk density at the 0-10 cm layer, with steeper slopes tending to have more compacted soils (p=0.0018). Steepness of slope also explained a significant portion of the variation in bulk density in the 30-40 cm layer in forests, and the relationship was again positive (r^2=0.4312, p=0.0057).

Although forest C pools were not correlated to any of the measured management variables, including distance from village and harvesting history, land owners were significantly more likely to have selectively harvested timber from their forest parcelas if the parcelas were closer to the village (P=0.0366). The most common timber species reported were *Anacardium excelsum*, *Hyeronima alchorneoides*, *Cedrela odorata*, *Guarea sp.*, *Platymiscium pinnatum*, and *Tabebuia rosea*.

None of the environmental or management variables in forest or pasture sites explained variation in diversity among sites. In agroforests, sites that were fallow for some period of time between when the forest was originally cleared and when the agroforest was established tended to have higher diversity (r^2=0.274, p=0.0373).

4. Discussion

4.1 Quantifying carbon stocks

The strength in our sampling design lies in its ability to capture heterogeneity in AGB with minimal sampling effort, and to examine variation among sites in addition to the variation among land-use types. By incorporating local geographic and ecological knowledge into the design we were able to increase our scientific accuracy without imposing barriers to local participation, and the local 'ownership' of the project was emphasized.

Our plot size allowed us to use a standard methodology across land-use types, as larger plots would not have fit in most agroforests. Other studies working on carbon stocks in smallholder land-use systems similarly have used small plots; for example, Fujisaka et al. (1998), working in the Amazon, used transects of 100 m^2 to compare forest, newly cleared and burned crop land, 2-3 year old fallows and pastures. Kotto-Same et al. (1997) used 100 m^2 plots to look at C stocks six different land-use types in Cameroon, including crops, secondary forest, and mature cacao forest. However, these two studies did not explicitly discuss the precision of their results for each land-use type, and the potential of applying their sampling methods for investor-targeted carbon inventories is therefore not clear. With paired plots of a total area of 0.14 ha, we have covered enough area to avoid a heavily skewed distribution of AGB among forest sites; tests for normality indicated that the data were normally distributed. Previous studies have suggested that plots smaller than 0.25 ha in size will often lead to a skewed distribution of AGB among plots because only a few of the plots will include a rare large tree within its boundary (Chave et al. 2003).

Although the variation around our means for above-ground biomass was high, a post-hoc sample analysis showed that precision could be reduced to +/- 10% of the mean with 95%

confidence with about 91 forest sites and 55 agroforest sites. In both forests and agroforests, the overwhelming contribution of trees $\geq$ 10 cm dbh to total AGB (and total error in AGB) suggests that the required increase in sampling effort need not be as great as the required increase in sample size. The results of the pilot project provide enough power to determine the average contribution of above-ground components other than trees to total AGB in these land use types, and eliminating the measurement of these components in additional sampling would greatly improve efficiency. In the case of pastures, the sample size analysis called for 1971 pasture sites. The reason for the high number of sites required for pastures is the patchy distribution of trees in pastures, and suggests that pairing our method with an alternative method that is better able to capture the presence of rare components such as trees and woody debris might be preferable (e.g. point-quadrat method (MacDicken 1997), use of aerial photographs for manual tree counts (R. Condit, pers. comm. 2004))..

On the other hand, a more realistic solution might be to determine the acceptable level of error in absolute terms, as a tonne of carbon is likely to be the unit in any carbon payment scheme. From this point of view, limiting the allowable error in pasture AGB to 0.37 Mg ha^{-1} (10% of the mean) does not make sense when the allowable error in forest measurements would be 32.6 Mg ha^{-1}. Instead, an absolute error of +/- 30 Mg C ha^{-1} might be allowed; this level of precision would require that we increase the total number of forest sites to 107, but we would not need to further sample the other land-use types.

Engaging the community in project planning and training local villagers to carry out the carbon inventories independently has brought a number of benefits to the project. First, the capacity for the community to participate in a carbon management scheme without being dependent on external expertise or aid, except perhaps in the area of data analysis, has been established. Team members also aquired skills in measurement, data collection, the use of geographical positioning systems, project planning and project execution that will last beyond the lifetime of the pilot project. Both of these aspects of the project provide examples of how the sustainable development objective of the CDM might be achieved (UNFCCC 1997). Second, team members were seen by the community as the local 'carbon experts', and were regularly approached for clarification on what exactly was "carbon". We believe that an understanding of the logic behind carbon management initiatives will be necessary for their successful implementation, and having a neighbour rather than a scientist explain technical concepts is likely to be much more intuitive to people with alternative worldviews (for related discussion see Alexiades 1996). Third, we were able to efficiently distribute sampling sites throughout the TC and to locate them in the field. Fourth, the knowledge of the local taxonomic experts allowed us to improve the scientific accuracy of our study without requiring a full-time field botanist.

This fourth point deserves particular emphasis, as our results confirm the value of calibrating allometric models for local wood density. We found that differences among models were reduced significantly by the correction, and across all models and land-use types our estimates of tree AGB were reduced by 11%. Although the mean wood densities of the land-use types we compared were not significantly different, other studies have shown that forests in different successional states have different average wood densities (Worbes 1997; Nebel et al. 2001); and in some cases differences in mean wood densities among forest types explain more of the variation in forest biomass than do differences in wood volume (Nebel et al. 2001). The wood density correction may therefore prove even more valuable when comparing forests and agroforests to secondary forests, which tend to be dominated by fast-growing pioneer species.

4.2 Carbon stocks in an inhabited landscape

One aspect of our study that differs from many studies of C storage that have recently been published is that the landscape that we sampled is inhabited and the forests are used by the community for timber, fuelwood, hunting, and artisan materials. For example, although the study sites of the Centre for Tropical Forest Studies in the Panama Canal Watershed (Chave et al. 2004) are likely to have experienced similar pressures in the past, the forests are currently protected from logging. More pristine research sites like Barro Colorado Island in Panama (Chave et al. 2003), La Selva Research Station in Costa Rica (Clark et al. 2002) and the Biodiversity of Forest Fragments site in Brazil (Nascimento and Laurance 2002) are even less disturbed. Given that most of the tropical forests of the world are inhabited, and these are certainly the forests that are under the most pressure, there is an urgent need to understand the relationships between the use of forests by adjacent communities, forest structure, and ecological processes such as carbon storage in soils. Although the use of the forests of the TC are much less intensive than in some areas, comparisons of the results of our study to those from more pristine sites may nevertheless help to elucidate these relationships.

Our estimate of AGB of trees $\geq$ 1 cm DBH in forests (340 Mg ha^{-1}) was higher than that for 40 one-hectare plots of late-secondary and primary forest in the Panama Canal Watershed (251.7 Mg ha^{-1}), where AGB was calculated using the same allometric model as in our study (Chave et al. 2004). However, the values are within one standard deviation of one another (Chave et al. 2004). The higher AGB values for our site may be explained by the large number of trees $\geq$ 70 cm DBH that we recorded: whereas Chave et al. (2004) warn that more than 15 trees $\geq$ 70 cm DBH ha^{-1} is unusual, when summed together our sites had an average of 19 trees $\geq$ 70 cm DBH ha^{-1}. Our randomized method for selecting sites was designed to avoid any subjective bias in sampling "majestic sites" (Sheil 1995; Phillips et al. 2002), and the normal distribution of AGB among our forest sites suggests that the mean of our 16 samples is not influenced by the presence of a few large trees in a small number of sites. Our site therefore appear to have a higher density of large trees than those sampled in the Canal Watershed.

Our estimates of the number of stems per ha in forest sites was also within one standard deviation of the average number of stems for the 40 CTFS plots (Chave et al. 2004). The average wood density of our forest sites (0.53 g cm^{-3}) was higher than the average site wood density of the 57 long-term monitoring plots in the Panama Canal watershed (0.499 g cm^{-3}), but again it was well within the range of average wood densities for these plots (0.32-0.61 g cm^{-3}; Chave et al. 2004).

Although landowners reported that they were more likely to harvest timber if their parcelas were closer to the village, we found no effect of reported selective logging or distance to village on AGB of trees per hectare. Timber harvesting in the TC appears to be of a low-enough intensity that we did not detect any effect in our pilot study. However, we did pass stumps when traveling between sample sites, and the loss of one large-diameter tree from a plot would certainly have a large effect on plot biomass (fig. 5a). More sampling might therefore detect an effect of selective logging on tree AGB.

Our estimates of the biomass of downed woody debris in forest sites (8.09 Mg ha^{-1}) is low compared to estimates from other Neotropical sites. Our result is significantly lower than the estimates of 27.86 Mg ha^{-1} , for a moist forest in the Central Amazon (Nascimento and Laurance 2002) and 46.3 Mg ha^{-1} for the wet primary forest of the La Selva reseach station in Costa Rica (Clark et al. 2002). However, our results are within one standard deviation of estimates of woody debris in a primary wet forest in the Los Tuxtlas region of Mexico (14 Mg ha^{-1}), and much closer to the estimates for woody debris in secondary forests (minimum age 20 years) in the same region (8.84 Mg ha^{-1}; Hughes et al. 1999). The quantity of downed woody debris would seem to be one component that would be particularly sensitive to fuelwood-collection by a nearby village, and forest owners in Ipetí explained that while timber harvesting and agriculture could only be carried out in one's own parcela, community members could collect fuelwood in any parcela in the TC. However, even though some of our sites were a three hour walk from the village, we failed to detect any relationship between the biomass of downed woody debris and distance from the village. This explanation for our low estimates of woody debris AGB therefore does not seem to be supported by the data.

Although we found that environmental variables did not account for any of the variation in carbon pools among our forest sites, other studies have similarly failed to link AGB to the underlying environment. For example, no relationship was found between AGB and soil type at La Selva forest, Costa Rica (Clark and Clark 2000), and a study of four lowland forests in Sarawak found no relationship between soil nutrient concentrations and AGB (Proctor et al. 1983). However, in Central Amazonia, AGB was shown to increase with soil fertility (Laurance et al. 1999). DeWalt and Chave (2004) attempt to clarify the uncertain effect of environmental gradients on forest structure by comparing the sensitivity of forest biomass to soil fertility across four Neotropical forests. In only one of the four forests they surveyed was there a relationship between AGB and soil fertility; DeWalt and Chave (2004) therefore conclude that there is little evidence for a constant response of forest biomass to soil fertility.

Given the variation in the definition of "agroforest" among studies, our results for this land use type are more difficult to compare. However, recent reviews of agroforestry systems allow us to place our results within the range of estimates for different types of tropical agroforestry systems. Kandji and Albrecht (2003) found the C sequestration potential of tropical agroforestry systems, including any land use system in which trees were deliberately retained or where trees were introduced with agricultural crops, pastures or livestock was between 12 and 228 Mg C ha^{-1}, with a median value of 95 Mg C ha^{-1}. For agroforestry systems in tropical America, the range was of 39-102 Mg C ha^{-1}. Our results for C storage in the agroforests of the TC of approximately 127 Mg C ha^{-1} (considering SOC to only 40 cm depth), are therefore at the high end of this range, which is not surprising given that the trees in the agroforests of Ipeti are the principal 'crop' of the land use system (as opposed to, say, cassava or cattle). Roshetko et al. (2002) surveyed homegardens of small-scale farmers in Indonesia that were comprised of a similarly diverse mix of fruit and timber tree species as the agroforests in Ipetí. Using the allometric model of Brown et al. (1997) and including SOC to a depth of only 30 cm, the 19 homegardens they surveyed had an average of 107 Mg C ha^{-1}. If we recalculate our total agroforest C stocks using the same allometric model for trees $\geq$ 10 cm DBH and excluding SOC for the 30-40 cm layer, we find an identical estimate of 107 Mg C ha^{-1}. The average age of the Indonesian agroforests was 13 years, while the average age of the agroforests in our study was 23 years. However, Roshetko et al. (2003) similarly note the mixed ages of trees in the agroforests they surveyed, suggesting that agroforest 'age' is perhaps a poor predictor of agroforest carbon stocks. A more complete analysis of agroforest C stocks in Ipetí would consider the length of the rotation of timber trees in the agroforests as well as the lifetime of the wood products that are harvested from the agroforests (Kandji and Albrecht 2003).

Our results for above-ground C stocks of pastures indicated that gains of 82 Mg C ha^{-1} and 210 Mg C ha^{-1} could be acheived by converting pastures to agroforests or forests respectively. Neither grazing intensity, nor alternative management practices for weedy herbs ("hierbas malas") seemed to affect the above- or below-ground biomass of the pastures we measured. SOC was significantly higher in pastures, which we believe reflects the compaction that occurs in more intensively managed (and less treed) land-use types. We therefore recommend that either bulk density and % C be considered separately when monitoring changes in soil characteristics, or that

SOC of intensively managed land-use types be corrected for the average bulk density of nearby forest sites. There was a weak trend of decreasing % C with pasture age, but the trend was not significant. However, our finding that % C and bulk density varied significantly among sites within land-use types suggests that initial variation in soil characters among sites may obscure any changes following land-use change. Such changes would be more easily detected if single sites could be monitored through time. Site-specific monitoring for carbon projects should overcome this problem.

4.3 Potential for C sequestration

The Clean Development Mechanism (CDM) defines forest as having a tree crown cover greater than 10%. This definition means that, in Ipeti, assisted regeneration of pasture to agroforest or forest would be eligible as a CDM project, but that restoration of agroforests to forests would not (Smith and Scherr 2001). However, the latter option would also be unlikely to appeal to agroforest owners in the first place, as agroforests provide multiple benefits including fruit, timber, artisan materials and medicines. Indeed, of all the landowners interviewed, only one indicated that he planned to abandon his agroforest in the future. However, that particular landowner had access to another agroforest that was closer to his home.

Economists argue that for payments for environmental services to work, the benefits perceived by the landowner for providing the service must outweigh the benefits of alternative land uses (Bishop and Landell-Mills 2002). A project focused on reforestation of pasture would therefore have to provide more incentives than landowners currently perceive for pasture creation. Among the sixteen pasture owners interviewed, nine indicated that they wished to maintain their pasture at its current size, and another three indicated that they planned to expand their pasture in the future. Two landowners said they would like to reforest part of their pastures, but maintain another part for cattle, while two landowners said they would like to reforest their pasture: one said he planned to establish a teak plantation when he could afford the seedlings, and the other said he was already in the process of allowing his pasture to regenerate naturally. A companion study to this one is currently working with the community of Ipeti to examine the economic and livelihood implications of alternative land uses and land management practices; the findings will help to develop carbon sequestration scenarios that will be socially feasible (Tschakert et al. in preparation).

Overall, the results of this study emphasise the technical feasibility of a CDM project based on restoration of pasture to either agroforest or forest. The benefits of reforestation to carbon storage in pasture are evident from the high coefficient of variation of tree AGB in pasture sites (table 8). For example, the addition of a single tree of 30 cm DBH to a site would increase the AGB of trees $\geq$ 10 cm DBH by an average of 95% for a pasture site, but by only 1% for a forest site and 3% for an agroforest site. Morphospecies richness was also positively correlated to number of stems within pastures, so reforestation of pastures would also be expected to increase the diversity of trees in these systems.

A reforestation program that uses agroforests or tree plantations as a transitional land use in the restoration of forests could be an interesting carbon management scenario with potential livelihood benefits. This type of reforestation model has been successfully experimented with in Veracruz, Mexico, where the sophisticated slash-and-burn agroforestry system of the Lacandon ethnic group has provided a model for the restoration of the local evergreen rainforests (Ramos-Prado et al. 2004). Under this scenario, the products and timber harvested from the 'transitional' agroforests and plantations in the short-term could provide financial support for landowners to develop other 'value-added' businesses or skill sets that would provide livelihood benefits in the long-term, and that would be less dependent on the intensive use of forests.

A further advantage of an agroforestry-focused carbon management project is the significant expertise in community surrounding this land use; in contrast to cattle ranching, the cultivation of multi-species agroforests in a traditional Emberá land use (Covich and Nickerson 1966; De Arauz 1970). There is also significant interest in the community for such a land-use, and since the initiation of the pilot project the community has hosted an agroforestry training course which was attended by a number of young farmers.

In their review of the enabling conditions for the adoption of agroforestry and small-scale plantations by local communities, Smith and Scherr (2001) highlight the following: (1) population densities need to be sufficiently high, and products from natural forests sufficiently scarce, to induce demand for planted forest products, and (2) local people have to have secure rights to harvest products of planted trees. The first condition could indeed be a constraint to the expansion of agroforestry in Ipeti: several landowners emphasized that the local market for their fruit was very limited. However, three landowners suggested that if they had access to a vehicle, they would be able to transport their produce to the markets of Panama City, which is approximately 2.5 hours by car. An interesting finding of our study was that agroforests from which oranges, avocados, or coffee were sold did not have less AGB than less intensively managed agroforests, and in fact had significantly more below-ground carbon. The higher levels of below-ground C may reflect the more intensive thinning that takes place in these agroforests; slash left on the agroforest floor it would be expected to contribute to soil C. Market-oriented agroforestry as carried out in Ipetí therefore does not appear to have disadvantages over less-managed agroforests in terms of C storage potential.

From a biodiversity perspective, agroforests contained significantly more native species than pastures, but also the most non-native tree species. The non-native species included *Citrus sinensis*, *Mangifera indica*, *Cocos nucifera*, and *Persea Americana*, all of which are widespread in

Panama and provide landowners with food products. Native species are of more interest for biodiversity conservation initiatives than non-native species, as native species are expected to have positive effects on other native species that are associated with them, such as specialist pollinators and seed predators. However, non-native species also contribute to the structural complexity of agroforests, which is certainly more likely to provide favourable 'matrix' habitat to wildlife than would pastures. The expansion of agroforests into areas now covered by pasture should therefore have concurrent benefits for biodiversity conservation.

5. Conclusions

The pilot project will continue over the coming year during which the community will debate the social feasibility of alternative carbon management and land use scenarios. Based on our results, expanding agroforestry into areas currently dominated by pasture appears to be an exciting option from a technical perspective and livelihood perspective. Agroforests of an average age of 23 years had about 82 Mg more C per ha than pastures. However, market-oriented agroforestery is likely to be of limited success without better access for agroforest owners to Panama City markets. In terms of protecting both existing carbon stocks and biodiversity, slowing the conversion of forest to pasture will have the greatest impact; in the short-term, this would avoid the release of approximately 210 Mg C ha^{-1}. Currently forest conservation is not included in CDM eligible-projects. Incentives for forest conservation are thus sorely needed.

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Chapter 4

Conclusion

Chapters 2 and 3 of this thesis offer several insights into the causes of and potential solutions to deforestation in the Neotropics.

Chapter 2 emphasises the strong correlation of roads and highways to deforestation in the Brazilian Amazon. This finding is in concert with previous empirical work, and suggests that current plans to expand infrastructure through remote areas of the Amazon basin could lead to significant increases in the deforestation of these areas. Although certain biophysical factors – such as a long dry season – may predispose land to clearing, I argue that biophysical factors that are unfavourable to agriculture are unlikely to act as deterrents to deforestation in the long-term as land becomes more scarce. In the context of the Brazilian Amazon, where vast areas of forest that are still inhabited only at very low population densities, Chapter 2 suggests that one of the most powerful means of controlling deforestation is by maintaining "passive barriers" to deforestation; in other words, forests that are difficult to access will be less likely to be deforested.

However, this type of solution puts more pressure on already-inhabited forest zones, and also provides no solutions for forested areas in other parts of the tropics that are inhabited and where forests are used by local people. In these situations, solutions to deforestation that aim to influence the perceived opportunities of people already in the land are much more likely to succeed. Chapter 3 shows the potential for success at the local-scale of market-based incentives that reward forest owners for the carbon storage and biodiversity conservation services that their forests provide.

Appendix 1

The soil fertility classification was based on the same EMBRAPA map of Amazonian soils as in the original study by Laurance and colleagues [41] for which the biophysical data was compiled. For the present study, soil units were reclassified into five main classes and four intermediate classes. The lowest class (1), comprising 8% of the basin, contained very infertile soils with no agricultural potential, such as quartz sands and podzols. The second class, comprising 13% of the total area, contained mostly distrophic (nutrient-poor) soils with strong physical restrictions, such as shallowness, high content of concretions, waterlogging, or plinthite. The third class, comprising 20% of the basin, contained highly weathered, acidic, distrophic soils with no severe physical restrictions other than light topsoil and clay enrichment with depth (Ultisols). The fourth class, comprising 21% of the area, contained highly weathered, acidic, distrophic soils with generally favourable physical properties (Oxisols). The fifth class, comprising 5.5% of the basin, contained relatively fertile soils that are nutrient-rich, not acidic and whose main restrictions are seasonal flooding (varzea soils) or waterlogging (Vertisols). This class also contained the Terra Roxa soils that are sought after for cocoa growing and other eutrophic soils.

\ppendix 2

pecies encountered in carbon inventory in the tierra collectiva of Ipetí-Emberá. Family, Genus and Species names are hortened to eight-letter code.

Local name	Family code	Genus code	Species code	Number encountered in forest sites	Number encountered in agroforest sites	Number encountered in pasture sites	Total number encountered
Mango	Anacardi	*Mangifer*	*indica*	0	197	7	204
Naranja	Rutaceae	*Citrus*	*sinensis*	0	180	0	180
Jira	Arecacea	*Socratea*	*exorrhiz*	140	0	0	140
Palma escoba	Arecacea	*Cryosoph*	*warscewi*	109	0	0	109
Cauchillo/Caucho	Moraceae	*Sorocea*	*affinis*	51	9	0	60
Coco	Arecacea	*Cocos*	*nucifera*	0	56	0	56
Guarumo	Cecropia	*Cecropia*	*sp*	20	33	1	54
Membrillo	Lecythid	*Gustavia*	*sp*	49	1	0	50
Cacao	Sterculi	*Theobrom*	*cacao*	1	48	0	49
Quiebra hacha	Sapindaceae	*Matayba*	*glaberri*	49	0	0	49
Cedro amargo	Meliacea	*Cedrela*	*odorata*	0	45	0	45
Guaba	Fabaceae	*Inga*	*nobilis*	3	41	1	45
Nance	Malpighi	*Byrsonim*	*crassifo*	0	40	2	42
Jobo	Anacardi	*Spondias*	*mombin*	12	19	10	41
Punula	Bombacac	*Quararib*	*asterole*	39	0	0	39
Guagara	Arecacea	*Sabal*	*mauritii*	30	3	2	35
Fruta paisana	no_id	*no_id*	*no_id*	33	0	0	33
Zapotillo	no_id	*no_id*	*no_id*	33	0	0	33
Carekidave	no_id	*no_id*	*no_id*	32	0	0	32
Aguacate	Lauracea	*Persea*	*american*	0	29	0	29
Espave	Anacardi	*Anacardi*	*excelsum*	10	18	0	28
Aguacatillo	Lauracea	*indet*	*indet*	25	2	0	27
Mandarina	Rutaceae	*Citrus*	*reticula*	0	25	0	25
Palo santo	Fabaceae	*indet*	*indet*	23	2	0	25
Roble	Bignonia	*Tabebuia*	*rosea*	1	23	0	24
Guabito	Fabaceae	*Inga*	*sp*	21	3	0	24
Cedro macho	Meliacea	*Guarea*	*sp*	23	1	0	24
Huesito	Rhizopho	*indet*	*indet*	24	0	0	24
Tamarindo/Zorro Macho	Fabeaceae:Ca	*Dialium*	*guianense*	23	0	0	23
Guacimo	Sterculi	*Guazuma*	*ulmifoli*	10	11	0	21
Sigua	Lauracea	*indet*	*indet*	16	2	1	19
Bejuco motete	no_id	*no_id*	*no_id*	19	0	0	19
Berba	Moraceae	*Brosimum*	*sp*	19	0	0	19
Guayabillo	Myrtacea	*indet*	*indet*	16	2	0	18
Sangre de gallo	Fabaceae	*Pterocar*	*belizensis*	16	0	1	17
Pifa	Arecacea	*Bactris*	*gasipaes*	0	16	0	16
Jordan	no_id	*no_id*	*no_id*	14	1	0	15
Mora	Moraceae	*Chloroph*	*tinctori*	8	5	1	14
Quira	Fabaceae	*Platymis*	*pinnatum*	10	3	1	14
Pierde	no_id	*no_id*	*no_id*	13	1	0	14
Cedro espino	Bombacac	*Pachira*	*quinata*	0	13	0	13
Caimito	Sapotace	*Chrysoph*	*cainito*	7	6	0	13
Bongo	Bombacac	*Ceiba*	*pentandr*	12	0	0	12
Chunga	Arecacea	*Astrocar*	*standley*	9	1	1	11
Fruta de loro	no_id	*no_id*	*no_id*	11	0	0	11
Achiote	Bixaceae	*Bixa*	*orellana*	0	10	0	10
Cuajao	no_id	*no_id*	*no_id*	9	1	0	10
Fruta de mono	Clusiace	*Garcinia*	*intermedia*	10	0	0	10
Garrapato	Chrysoba	*indet*	*indet*	10	0	0	10
¯ ıaba macho	Fabaceae	*Inga*	*spectabi*	0	9	0	9
nbito	no_id	*no_id*	*no_id*	9	0	0	9
ıfé	Rubiacea	*Coffea*	*arabica*	0	8	0	8
ıanabana	Annonace	*Annona*	*muricata*	0	8	0	8
ya	Annonace	*Unonopsi*	*panamens*	8	0	0	8

ocal name	Family code	Genus code	Species code	Number encountered in forest sites	Number encountered in agroforest sites	Number encountered in pasture sites	Total number encountered
laranjillo	no_id	*no_id*	*no_id*	3	4	0	7
Cuipo	Bombacac	*Cavanill*	*platanif*	5	2	0	7
Cuadra	no_id	*no_id*	*no_id*	7	0	0	7
Mentol	Fabaceae	*Myroxylo*	*balsamum*	7	0	0	7
Pinuguillo	no_id	*no_id*	*no_id*	7	0	0	7
Popochira	no_id	*no_id*	*no_id*	7	0	0	7
Poroporo	Cochlosp	*Cochlosp*	*vitifoli*	2	0	5	7
Mamon	Sapinace	*Melicocc*	*bijugatu*	0	6	0	6
Totumo	Bignonia	*Crescent*	*cujete*	0	6	0	6
Guayaba	Myrtacea	*Psidium*	*guajava*	1	5	0	6
Tachuelo	Rutaceae	*Zanthoxy*	*sp*	6	0	0	6
Mamey	Sapotace	*Pouteria*	*sapota*	0	5	0	5
Teca	Verbenac	*Tectona*	*grandis*	0	5	0	5
Guabo pelu	Fabaceae	*Inga*	*sp*	0	4	1	5
Balso	Bombacac	*Ochroma*	*pyramida*	4	1	0	5
Hierba de montana	no_id	*no_id*	*no_id*	5	0	0	5
Midala	no_id	*no_id*	*no_id*	1	0	4	5
Santa maria	Clusiace	*Calophyl*	*longifol*	5	0	0	5
Palma real	Arecacea	*Attalea*	*butyrace*	0	4	0	4
Jagua	Rubiacea	*Genipa*	*american*	2	2	0	4
Ortiga	Urticace	*Urera*	*baccifer*	2	2	0	4
Laurel/Laureño	Boraginaceae	Cordia	alliodora	4	0	0	4
Palo bejuco	no_id	*no_id*	*no_id*	2	0	2	4
Zorro	Anacardiaceae	Astronium	graveolens	4	0	0	4
Ciruelo	Anacardi	*Spondias*	*purpurea*	0	3	0	3
Maranon	Anacardi	*Anacardi*	*occident*	0	3	0	3
Nazareno	Bignonia	*Jacarand*	*copaia*	1	2	0	3
Chocolate silvestre	Sterculi	*Herrania*	*purpurea*	3	0	0	3
Guabito de monte	no_id	*no_id*	*no_id*	3	0	0	3
Macano	Fabaceae	*Diphysa*	*robinioi*	3	0	0	3
Zapatero	Euphorbi	*Hyeronim*	*alchorne*	3	0	0	3
Caoba	Meliacea	*Swieteni*	*macrophy*	0	2	0	2
Limon	Rutaceae	*Citrus*	*limon*	0	2	0	2
Majagua	Tiliacea	*Heliocar*	*american*	0	1	1	2
Panama	Sterculi	*Sterculi*	*apetala*	0	1	1	2
Aguacatillo de monte	Rubiacea	*Tocoyena*	*pittieri*	2	0	0	2
Cigarillo	Fabaceae	*Schizolo*	*parahyba*	2	0	0	2
Fruta de moracho	no_id	*no_id*	*no_id*	2	0	0	2
Fruta de tucan	no_id	*no_id*	*no_id*	2	0	0	2
Majaguillo	Tiliacea	*Trichosp*	*galeotti*	2	0	0	2
Malagueto	Annonace	*Xylopia*	*sp*	2	0	0	2
Maquenqe	Arecacea	*Oenocarp*	*mapora*	2	0	0	2
Nispero	Sapotace	*Manilkar*	*sp*	2	0	0	2
Palo sapo	no_id	*no_id*	*no_id*	2	0	0	2
Pepita	Lythrace	*Lafoensi*	*punicifo*	2	0	0	2
Borojo	Rubiacea	*Borojoa*	*panamens*	0	1	0	1
Fruta de pichilingo	no_id	*no_id*	*no_id*	0	1	0	1
Maranon curazao	Myrtacea	*Syzygium*	*malaccen*	0	1	0	1
Pino	no_id	*no_id*	*no_id*	0	1	0	1
Amargo-Amargo	Fabaceae	*Vatairea*	*erythroc*	1	0	0	1
Amarillo	Combreta	*indet*	*indet*	1	0	0	1
Caracho	no_id	*no_id*	*no_id*	0	0	1	1
Ceibo	Euphorbi	*Hura*	*crepitan*	1	0	0	1
Ceniza pono	no_id	*no_id*	*no_id*	1	0	0	1
Coquillo	Euphorbi	*Jatropha*	*curcas*	1	0	0	1
Corotu	Legumino	*Enterolo*	*cyclocar*	1	0	0	1
or de verano	no_id	*no_id*	*no_id*	1	0	0	1
jolil	Fabaceae	*indet*	*indet*	0	0	1	1
uta de conejo	no_id	*no_id*	*no_id*	1	0	0	1
arumo macho	Cecropia	*Pourouma*	*bicolor*	1	0	0	1

ocal name	Family code	Genus code	Species code	Number encountered in forest sites	Number encountered in agroforest sites	Number encountered in pasture sites	Total number encountered
;uayacan	Bignonia	*Tabebuia*	*sp*	1	0	0	1
Higo	Moraceae	*Ficus*	*carica*	1	0	0	1
Jagua macho	Rubiacea	*Randia*	*armata*	1	0	0	1
Kerosin	no_id	*no_id*	*no_id*	1	0	0	1
Mata palo	Moraceae	*Ficus*	*obtusifo*	1	0	0	1
Olivo	Euphorbi	*indet*	*indet*	1	0	0	1
Peronil	Fabaceae	*Ormosia*	*sp*	1	0	0	1
Sinba	no_id	*no_id*	*no_id*	1	0	0	1
Tabaquillo	Verbenac	*Aegiphil*	*anomala*	1	0	0	1
Zorrillo	no_id	*no_id*	*no_id*	1	0	0	1

Appendix 3

Following this page are waiver letters from the co-authors of Chapter 2 giving me permission to include the manuscript in my thesis.

The letter of permission from Scott Bergen is expected to arrive by fax on August 31st to the Department of Biology. It will be attached here.