MULTI-TEMPORAL ANALYSIS OF LAND-COVER CHANGE AND FOREST FRAGMENTATION PATTERN IN SOUTHEASTERN BRAZILIAN AMAZON USING GIS AND REMOTE SENSING (1986-2005)

By

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To the joy of my life: My daughters (Karol and Alana), my husband (Abib), and my parents (Roberto and Carmélia), always in my heart.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

BLA	Brazilian Legal Amazon
CBERS	Satélite Sino Brasileiro de Recursos Terrestre
CIPEC	Center for the Study of Institutions, Population, and Environmental Change
DN	Digital Number
DGI	Image Generation Division
ETM	Enhanced Thematic Mapper
F- NF	Forest-Non Forest
FUNTAC	Fundação de Tecnologia do Acre
GCP	Ground Control Point
GLCF	Global Land Cover Facility
GO	Governmental organization
HSD	Human and Social Dynamic
IBGE	Brazilian Institute of Geography and Statistics
IDL	Interactive Data Language
IMAZON	Institute of Man and Environment in the Amazon
INPE	Brazil's National Institute of Spatial Research
IMAC	Environmental Institute of Acre
ISOSEG	Interactive Self-Organizing Data Analysis Technique
ISODATA	Interactive Self-Organizing Data Analysis Techniques
LAGEOP	Laboratório de Geoprocessamento
LSMM	Linear Spectral Mixture Model
LULCC	Land Use and Land Cover Change
MAP	Madre de Dios (Peru), Acre (Brazil) and Pando (Bolívia)

NASA	National Aeronautics and Space Administration
NGO	Non-Governmental Organization
NSF	National Science Foundation
PCI	Principal Component Analysis
PES	Payments for Environmental Services
PRODES	Satellite Image Monitoring Program of Brazilian Amazon Forest
REDD	Reduced Emissions from Deforestation and Degradation
RMS	Root Mean Square
TM	Thematic Mapper
UFAC	Federal University of Acre
UF	University of Florida
UM	University of Maryland

Abstract of Dissertation Presented to the Graduate School
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MULTI-TEMPORAL ANALYSIS OF LAND-COVER CHANGE AND FOREST FRAGMENTATION PATTERN IN SOUTHEASTERN BRAZILIAN AMAZON USING GIS AND REMOTE SENSING (1986-2005)

By

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Tropical deforestation is a key process influencing land cover dynamics as well as global climate change. While our understanding of the drivers and consequences of deforestation has grown rapidly in the past two decades, there remain significant debates concerning its estimation, new calls for research on its causation, and growing doubts about its mitigation via public policies. This dissertation therefore takes up these questions by bringing together remote sensing of land-cover, theoretical frameworks for understanding causation behind land cover change, and insights from landscape ecology about forest fragmentation in order to evaluate the estimation, causation, and spatial-temporal aspects of deforestation. I focus on the case of Acre, located in the Brazilian Amazon. Acre is a useful case study because it has incurred deforestation in recent years, but the spatial patterns and temporal dynamics vary considerably. Acre is witness to two key causal factors behind deforestation, new infrastructure and changes in land tenure. Acre also became the focus for a debate within Brazil over competing sources of deforestation estimates as a means of evaluating public policies for sustainable development. Finally, Acre is a global leader in the design and implementation of payments for ecosystem services programs that seek to reduce environmental degradation. The heart of this dissertation resides in three analytical

papers. The first takes up the question of the importance of the processing and classification of remote sensing data as they may affect deforestation estimates. I compare three sources of deforestation estimates for Acre and systematically evaluate their processing protocols. The estimates vary considerably, and land cover classification emerges as the main explanation. This bears implications for selecting processing protocols for deforestation estimation to evaluate public policies. The second paper focuses on the issue of the relative importance of infrastructure and land tenure for deforestation trajectories. Both causal factors have large literatures and previous empirical work, but there has been less attention to whether these factors operate independently of each other. I focus on eastern Acre, where the Inter-Oceanic Highway has been paved over time across a mosaic of diverse land tenure types. This permits comparisons of deforestation estimates over time for an array of lands which received paving at different times, exist at different distances from the highway, and have different land tenure rules. The findings show that accessibility and tenure both exert important effects on deforestation, and that the interaction between infrastructure and tenure is not strong as sometimes supposed. However, there is also evidence of rule-breaking, which also bears implications for theory and policy. The third and final paper focuses on the issue of the spatial pattern of deforestation. I draw from landscape ecology, which makes specific predictions about forest fragmentation and ecological degradation, to pursue an analysis of fragmentation over time in eastern Acre. I focus on Directed Settlement Projects, or PADs, which have the same tenure rules, but which in eastern Acre have diverse local road networks that may yield very different fragmentation patterns. I compare fragmentation over time in the PADs in terms of a suite of pattern metrics, and observe some differences but also important similarities in their landscape mosaics. These findings bear implications for the question of whether to target payments for ecosystem services programs at specific types of lands with particular fragmentation patterns.

CHAPTER 1
INTRODUCTORY REMARKS

Tropical deforestation is among the global environmental issues that have received great attention by the scientific community in the last decade due to the huge implications for the world's terrestrial carbon cycling and climate regulation (Rayner and Malone 2001). It will undoubtedly continue to be a central international environmental issue in the coming years, particularly as a source of carbon emissions related to global climate change. Tropical forests account for slightly less than half of the world's forest area, yet they hold about as much carbon in their vegetation and soils as temperate-zone and boreal forests combined. Trees in tropical forests hold, on average, about 50% more carbon per hectare than trees outside the tropics. Thus, deforestation of a given land area will generally cause more carbon to be released from the tropical forests than from forests outside the tropics.

The rapid destruction, degradation and impoverishment of tropical forests through deforestation is considered a major source of greenhouse gases such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_20). These gases play an important role in global climate, and their emissions through deforestation contributes to climate change. Tropical deforestation and forest degradation released 0.5 to 2.4 billion tons of carbon each year during the 1990s (Houghton 2005a), and was therefore 0.8 to 2.8% of the annual worldwide human-induced emission of carbon to the atmosphere.

Recently, it has been estimated that 20% of greenhouse gases emissions came from land-use/land-cover-change (LULCC), principally tropical forest conversion for human land-uses (Gullison et al. 2007; Boucher 2008). However, carbon sequestration by forests and non-forest vegetation plays an important role to help offset carbon emissions. The amount of carbon held in trees is 20-50 times higher than in cleared land. Also, changes in carbon stocks vary with the

type of land use, like the conversion of forest to croplands or pasture, with the type of ecosystem (tropical moist or dry forest), and with the tropical region, if we are thinking about South America, Africa or Asia.

Previous research also has indicated that large-scale deforestation not only result in climate change, but also in the loss of biological diversity, changes in hydrological cycles, and soil erosion and degradation (Houghton et al. 1991; Skole and Tucker 1993). After deforestation, regeneration of vegetation is common and the forest habitats become fragmented, turning the landscape into a mosaic of patches of successional forest and agricultural lands. Forest fragmentation exposes remaining forests to disturbances along forest/non-forest edges. As new forest edges are formed, remnant forests become increasingly affected by disturbances. In the interiors of forest fragments, tree mortality can rise and result in biomass collapse. This in turn results in carbon emissions (Laurance et al. 1997; Laurance et al. 2000, 2008). Light penetration through a more open canopy and the increased amount of woody litter can render these forests more susceptible to fire. Therefore, as forests become more fragmented, the risk of forest fires increases, which in turn also increases carbon emissions. Variation in biomass can be a good indicator of changes in these processes, as biomass governs the potential carbon emission that could be released to the atmosphere due to deforestation. A number of studies have provided useful approaches for estimating biomass, and thus carbon, as a basis for estimating carbon emissions related to deforestation (Achard et al. 2004; Hese et al. 2005; Houghton 2005).

According to the latest IPCC report (2007), human activities, primarily the burning of fossil fuels and clearing of forests, have greatly intensified the natural greenhouse effect, causing global warming. This is caused primarily by increases in “greenhouse” gases such as carbon dioxide which is the most significant one. By clearing forests to support agriculture, we are transferring carbon from living biomass into the atmosphere and consequently affecting climate

change. This issue has therefore increased international interest in approaches to reduce emissions from deforestation and forest degradation in developing countries (Metz et al. 2007, Ramankutty et al. 2007).

Such discussions raise questions about the causation behind deforestation, as well as policies to modify that causation and reduce deforestation. There is a large literature on the causes behind deforestation and other forms of LULCC (Lambin and Geist 2006, Gutman et al. 2004; Wood and Porro 2002, Angelsen and Waimowitz 1999). Pre-eminent frameworks for understanding LULCC feature multi-step causation from distant to intermediate to proximate factors. Distant factors include public policies; intermediate factors include infrastructure; and proximate determinants typically highlight land use decisions.

Infrastructure is one determinant of LULCC that has received considerable attention (Perz et al. 2007; Andersen et al. 2002; Fearniside 2002; Geist and Lambin 2001, Pfaff 1999). Infrastructure upgrades improve accessibility to land, rendering land use more profitable. This results in expanded deforestation and land use for market production beyond that needed for subsistence alone. Given that infrastructure often results in accelerated deforestation, there is debate about the wisdom of new infrastructure projects in forested regions such as the Amazon.

Another key issue concerning the causation behind infrastructure concerns institutions. Institutional arrangements define the “rules of the game” for human action, including resource management, such as deforestation and land use. In particular, land tenure rules are intended to define who gets access to land and what they may do with it. Tenure insecurity has been held out as an explanation for rapid deforestation (Alston et al. 1999), but tenure rules may vary and greatly determine whether deforestation is even allowable on specific types of lands.

Infrastructure and institutions are likely to interact in important ways to affect LULCC. Whereas lands closer to infrastructure are more valuable and thus more likely to be deforested, if

there are adjacent lands at equal distances from the same infrastructure but with different tenure rules, they may exhibit very different land cover. This may in turn become more evident over time as land cover trajectories develop.

The negative consequences and complex causation of LULCC have driven the search for public policies to reduce deforestation. Deforestation estimates have been featured in discussion of the design and implementation of payments for ecosystem services (PES). PES programs seek to motivate conservation of forests and other habitats that store carbon and provide other ecosystem services by pricing those services and paying land stewards who maintain them. Carbon PES programs in particular have focused on conservation of carbon in standing biomass in order to avoid carbon emissions that contribute to climate change.

In this context, REDD (Reduced Emissions from Deforestation and Degradation) emerged as an important carbon credit regime. REDD was consolidated within the United Nations Framework Convention on Climate Change (UNFCCC) at Bali in December 2007 for the post-2012 period. REDD therefore could compensate tropical countries for their nation-wide reduction in emissions from deforestation and forest degradation. It is supposed that REDD works as an opportunity for avoiding the worst consequences of global warming while generating enormous benefits for biodiversity conservation and sustainable development (Hall 2008; Boucher 2008).

The logic of REDD and other PES programs relies on several things, among them accurate estimates of deforestation before and during PES programs. In the case of REDD, carbon payments are made based on a reduction in deforestation rates from a baseline to the project period. Quantifying the spatial and temporal dynamic of deforestation and forest fragmentation over time to monitor global carbon pools and fluxes is therefore necessary. This challenge requires accurate geographic information to map land-cover change (Dixon et al. 1994;

McGuire et al. 2001) to support estimation of biomass change (Lu 2005; Lu et al. 2005, Houghton 2005, Achard et al. 2004).

Remote sensing (RS), alone or integrated with GIS, and field data, can be a valuable source of land cover and biomass data, as it provides a representation of the Earth's surface that is spatially continuous and highly consistent. RS can accurately obtain information of land cover conversion rapidly, cheaply and over a range of spatial and temporal scales. In addition, insights from landscape ecology about the spatial pattern of LULCC and the tools of geographic information systems (GISs) permit an evaluation of forest fragmentation. I therefore draw on land science frameworks for understanding the causation behind deforestation, RS data and protocols for observing deforestation, and spatial analysis of forest fragmentation to pursue a three-part analysis of land cover change, focusing on the case of the Amazon.

Specifically, I take up the State of Acre, located in the southwestern Brazilian Amazon. Acre is an interesting study case because it has incurred deforestation in recent decades. Popular explanations for deforestation in Acre highlight infrastructure projects there, notably recent initiatives to pave the Inter-Oceanic Highway (BR-317 in Brazil) and the BR-364 highway. At the same time, Acre has been a significant policy innovator, with leading examples of new land tenure types such as extractive reserves, which have specifically stated deforestation limits. Land tenure in Acre is now highly diverse, even along highways, which makes Acre a useful case study for an evaluation of differences in deforestation.

In the context of these innovative policies, Acre has also become the focus of debate over deforestation estimates. Despite many innovative policies to reduce deforestation, some estimates have indicated acceleration in forest clearing in Acre. This led to polemics over the question of the RS data sources and processing protocols. Acre thus becomes a very useful case study for comparing deforestation estimates from different sources.

Acre is also a useful study region for an evaluation of forest fragmentation. Landscapes in Acre also exhibit diverse spatial patterns of deforestation. This likely reflects the spatially varying effects of highways and land tenure. But even within categories of land tenure rules, the design of local road networks also varies. This raises interesting questions about the design of rural agricultural settlements and the ensuing fragmentation patterns. If local road networks do lead to differing patterns of fragmentation, that in turn bears implications for PES programs, which may be more effective in some settlements than others. More isolated fragments have less ecological value, and PES investments in their conservation may be less effective in securing forest biomass.

This dissertation therefore pursues a three part research agenda concerning deforestation: 1) understand sources of differences in deforestation estimates 2) evaluate the importance of infrastructure and land tenure for deforestation trajectories over time; and 3) compare forest fragmentation in settlements with differing road networks to inform strategies for PES programs. Each paper can be read as an independent document, which addresses a unique aspect of the overarching study. In the remainder of this introductory chapter, I discuss each of these analyses.

Variation in Estimates of Deforestation: A Comparative Assessment of Three Remote Sensing Protocols for the Case of Acre, Brazil

The first paper is entitled "Variation in Estimates of Deforestation: A Comparative Assessment of Three Remote Sensing Protocols for the Case of Acre, Brazil." This paper compares three remote sensing protocols used for assessing deforestation estimates in southeastern Brazilian Amazon. I focus on differences in the processing protocols and classification decisions that may cause deforestation estimates differences. My overall research question in this chapter is: How do differences in image processing protocols affect deforestation estimates? All of the estimates are based on Landsat images with comparable spatial resolutions.

Since these three different protocols follow somewhat distinct protocols for each processing step, making each step a potential source of differences in the resulting deforestation estimates, I then explore how these differences affect deforestation estimates. The main goal of this chapter is therefore to evaluate specific steps in remote sensing methodology to identify factors that account for differences in the resulting deforestation estimates. The chapter then presents available estimates of deforestation in Acre from each source at the municipal level and over time, which permits spatial and temporal comparisons. I intend to submit this paper to *International Journal of Remote Sensing*.

Land Tenure, Road, and Deforestation Patterns in Southeast State of Acre – Brazil

The second paper is entitled "Land Tenure, Road, and Deforestation Patterns in Southeast State of Acre – Brazil." I investigate the effect of paving status, distance to highway and land tenure type on deforestation estimates over time. The overall question in this chapter is: To what extent do deforestation estimates vary across highway paving status, distance to the main road and different land tenure models? I employ a time series analysis of Landsat Thematic Mapper (TM) imagery from 1986, 1991, 1996, 2000, and 2005 to evaluate the spatial and temporal distribution of deforestation. In this chapter, I focus on Acre state, since Acre is a very useful study case due to its incurred high-profile infrastructure investments as well as it is being policy laboratory for several innovative land tenure models, including some which have since diffused to other parts of Brazil. I compare 5 different models of settlement; these settlement categories have different land use policies and/or land-use strategies that vary from one settlement model to another. I first evaluate land deforestation due to paving status and distance to mean road network, then I evaluate how this land distribution by paving status and main road network varies according to different settlement models. The analysis involves several steps and explicitly considers whether the effects of paving and distance from highways remain when

simultaneously considering land tenure rules. I intend to submit this paper to the *Journal of Land Use Science*.

Measurement and Characterization of Patterns of Forest Fragmentation in the Southwest Amazon: Satellite Data Analysis from 1986 to 2005

The third paper is entitled "Measurement and Characterization of Patterns of Forest Fragmentation in the Southwest Amazon: Satellite data analysis from 1986 to 2005." This part of the analysis deals with the measurement and characterization of spatial patterns as well as temporal dynamics of forest fragmentation. This paper combines LULCC data obtained from satellite images, with insights from landscape ecology concerning the ecological viability of habitat fragments, and measures of landscape pattern metrics, in order to understand spatial patterns and temporal dynamics of forest fragmentation. The analysis focuses on a specific land tenure type in Brazil, the Directed Settlement Project (PAD). In Acre, PADs have diverse road networks, which may generate very different fragmentation patterns and dynamics. The overall question in this chapter is: Do differing road networks result in different patterns and dynamics of forest fragmentation? PADs in eastern Acre permit systematic comparisons, since they are located at different distances from the state capital, Rio Branco, and PADs both near and far from the capital have distinct road networks. In this chapter forest and non-forest classified images were input into fragstats and metrics generated at landscape and class levels. Of particular interest is my multi-temporal data, which permit a dynamic analysis of spatial pattern metrics among the PADs. I intend to submit this manuscript to *Landscape Ecology*, with *Remote Sensing* as a back-up plan.

Importance of the Study

Together these three papers contribute to a broader understanding of land-cover change and fragmentation dynamics associated with road networks in Western Amazonia. Theory and

methods from landscape ecology, land change science, remote sensing and GIScience contribute to develop this research in order to better analyze and understand spatial patterns and temporal dynamics of deforestation and habitat fragmentation. The approach outlined in the dissertation reveals how deforestation estimates have been controlled by different processing protocols, different time periods, paving status, distance to main road and different geographic spaces. The results of this research also have application for government agencies and NGOs in Western Amazonia interested in sustainable development and in the design and implementation of payment for ecosystem services (PES) programs.

CHAPTER 2
VARIATION IN ESTIMATES OF DEFORESTATION: A COMPARATIVE ASSESSMENT OF THREE REMOTE SENSING PROTOCOLS FOR THE CASE OF ACRE, BRAZIL

Summary

Many deforestation estimates have been derived from the Landsat platform in the past 30 years. More recently, estimates have also been produced from other orbital platforms, or using distinct methods of image processing and classification. As a result, there is often a diversity of estimates of LULCC available for high-profile study regions. Differences among deforestation estimates have taken on growing importance. Deforestation is increasingly seen as a metric for evaluating the effectiveness of environmental policies, and different estimates can be politicized by groups with interest in higher or lower estimates. Further, clean development mechanisms involving forest conservation have advanced toward implementation but require "accurate" estimates of forest cover with which to determine environmental service payments. Such issues are at play in many regions, notably the Brazilian Legal Amazon (BLA), where deforestation has proceeded rapidly, resulting in biodiversity loss and carbon emissions. This chapter therefore compares deforestation estimates in the Amazon using data from multiple remote sensing studies. The main goal is to evaluate specific steps in remote sensing methodology to identify factors that account for differences in the resulting deforestation estimates. The comparison focuses on deforestation estimates from three sources: 1) INPE, Brazil's National Institute of Spatial Research, which is responsible for producing official deforestation estimates for the BLA; 2) IMAZON, which produced its own deforestation estimates for Acre and 3) The National Science Foundation-funded Human and Social Dynamics project at the University of Florida (UF NSF project) which produced independent estimates of LULCC. The analysis shows differences in deforestation estimates; while there are many possible sources of such differences, in this analysis estimates vary primarily due to definitions of land cover classes.

Introduction

Technological advances in remote sensing, especially in the form of earth observing satellites, have made it easier for the scientific community to analyze the spatial extent of human impact on the environment, as well as naturally occurring environmental changes. Remote sensing enables large-scale observation of areas that would be inaccessible or otherwise difficult to access, making it applicable as a tool for monitoring Land Use and Land Cover Change (LULCC), since such changes are more difficult to quantify over large land areas using field methods of data collection. Satellite images play a very important role in the analysis of LULCC because they can cover large land areas with comparable data over time, both of which are important when studying forest changes (Yuan 2008; Dwivedi et al. 2005; Wood and Skole 1998). Up to date remote sensing data can be obtained across a range of spatial and temporal scales at a reasonable cost. Some software and satellite images can be downloaded from the internet for free. As a result, numerous universities, research centers, governmental organizations (GO's) and non-governmental organizations (NGO's) around the world are conducting LULCC studies based on remote sensing data that can help understand LULCC dynamics.

However, there remain questions about the reliability and comparability of remote sensing data, since the characteristics of orbital platforms, processing protocols, and classification algorithms all vary, which may affect estimates of LULCC. This is problematic, since varying estimates of LULCC bear ramifications for assessments of land use, productivity and degradation, which in turn may inform policies in various sectors such as agriculture, forestry and environment. In contexts where LULCC is politically sensitive, varying estimates of LULCC can complicate public discourse and policy making. In Brazil, deforestation has become a central environmental question concerning the Amazon. Policy initiatives to reduce the rate of tropical deforestation became especially relevant in Brazil where forest loss is responsible for

three-quarters of national carbon emissions, and contribute significantly to global warming (Stern 2006).

It is therefore environmentally, economically and politically important to ensure clarity regarding remote sensing protocols when presenting LULCC estimates. More specifically, it is crucial to identify specific sources of differing estimates of LULCC, which may result from decisions made at various steps of satellite image processing and classification. A comparison of methodologies to estimate LULCC in a given area over a specific period of time can be very important to evaluate the consistency of data inputs for policy.

Many deforestation estimates have been derived from the Landsat platform in the past 30 years. More recently, estimates have also been produced from other orbital platforms, or using distinct methods of image processing and classification. As a result, there is a diversity of estimates of LULCC available for high-profile study regions. One such region is the Brazilian Legal Amazon (BLA), where deforestation has proceeded rapidly, resulting in biodiversity loss and carbon emissions (Keller et al. 2009).

Deforestation estimates have become a metric relevant to the evaluation of various types of policies, not only those tied to various environmental impacts, but also others related to priorities such as economic growth. Whereas deforestation entrains environmental change, it is also instrumental for agricultural production and national development. Interest groups in support of one or another of these policy goals thus cite deforestation estimates. But with the availability of different deforestation estimates, different groups cite different estimates to support contrasting policy priorities. Insofar as environmental groups seek to reduce deforestation, they may cite higher deforestation estimates in order to demand environmental enforcement; and insofar as economic development interests seek further expansion of production, they may cite lower deforestation estimates to argue that the damage is less than

supposed and there remains room for growth. The result is contention among policy interest groups where deforestation estimation is a politically sensitive issue, regardless of the data source and processing protocols employed.

Such contention does not exempt the state, which encompasses many agencies with distinct mandates. Government on multiple levels is not only involved in data production including estimation of land cover change, but is also charged with formulating and implementing environmental and economic policies, including agricultural policy. Indeed, beyond interest groups, there is considerable potential for debate among governmental entities over deforestation estimates as they bear distinct ramifications among agencies with different mandates and operating on different levels (national, state or local).

For example in 2003, Brazilian media brought up a report published by the National Institute for Space Research (INPE) that highlighted rapid deforestation in the state of Acre (Veja 2003). The Government of Acre, which called itself the "Forest Government" due to its support of forest-based development, recognized the political threat posed by the high deforestation estimate, and took exception to the publication. Governor Jorge Viana challenged the deforestation and demanded an audit of the data. This resulted in a review of the methodological protocol, which revealed that the classification contained an error such that bamboo forests were misclassified as deforestation, resulting in high estimates of forest loss.

The 2003 debate over deforestation in Acre was not a unique event. In 2007, another national publication in Brazil (Veja 2007) again stimulated political debate over deforestation and government policy in Acre (Veja 2007). The news publication used a study commissioned by the Government of Acre to the Institute of Man and Environment in Amazonia - IMAZON (Souza et al. 2006). The IMAZON study showed that in the first six years of the Viana administration's Forest Government, the rate of deforestation in Acre tripled and reached 995

km² in 2004. The national publication used the IMAZON study to discredit the Acre's Forest Government. This reporting generated many headlines in Brazilian media and led to debate over deforestation estimates from different sources.

Beyond the politics of deforestation, accuracy in remote sensing is also important for LULCC estimates as an input for carbon estimation for application in clean development mechanisms, particularly programs involving payments for environmental services (PES). Carbon PES programs seek to provide incentives to avoid or reduce carbon emission from deforestation (Hall 2008). The establishment of carbon PES programs has numerous requirements, among which are reliable estimates of forest cover. The extent of forest cover and clearing, including measurement of rates of forest loss over time, are crucial to the calculation of the financial resources to be transferred to landholders. The extent of reductions in forest loss, and thus the amount of carbon emission avoided, provides the basis for determining the size of financial payments. However, if there are multiple sources of LULCC data, estimates of forest loss may vary, along with the carbon payments implied.

This situation thus begs questions about the validity of different remote sensing estimates of LULCC, and which set of estimates should be employed in PES programs. Understanding the different methods and procedures to produce deforestation estimates using remote sensing data is very important for PES programs, including Reduced Emissions from Deforestation and Degradation (REDD). Technically speaking, the preferable estimates should be those found to be most methodologically sound. However, given the economic ramifications of LULCC estimates for PES, different interests may prefer a given LULCC estimates for financial reasons. Further, the LULCC estimates adopted may in turn influence the incentive to avoid deforestation. If LULCC estimates implying smaller carbon PES adopted, participating landholders may be less enthusiastic about avoiding deforestation.

This chapter therefore compares deforestation estimates in the Amazon using data from multiple remote sensing studies. My goal is to evaluate specific steps in the remote sensing methodology in order to identify factors that account for differences in the resulting deforestation estimates. The comparison focuses on deforestation estimates from three sources: 1) INPE, Brazil's National Institute of Spatial Research, which is responsible for producing official deforestation estimates for the Brazilian Legal Amazon - BLA; 2) IMAZON, which produced its own deforestation estimates for Acre that became the focus of the 2007 controversy; and 3) a National Science Foundation-funded Human and Social Dynamics project at the University of Florida which produced independent estimates of LULCC in Acre and other parts of the southwestern Amazon. I will focus on data from all three sources for the Brazilian state of Acre. I chose Acre as there are multiple estimates available for land cover over time in this area, and because it has been the focus of previous controversies over deforestation estimates. I first review prior work on remote sensing of deforestation in the BLA, before focusing on the three data sources I employ. I then review the methodological protocols of each data source, noting their similarities and highlighting their differences. Each protocol involves multiple steps to derive deforestation estimates, and given differences in protocols among the studies at hand, each step thus constitutes a source of explanation for differing deforestation estimates. The chapter then presents available estimates of deforestation in Acre from each source at the municipal level and over time, which permits spatial and temporal comparisons. The analysis focuses on specific years for which there are data available from the three sources. These years permit direct comparisons in deforestation estimates, and identification of differences, which I then relate back to the methodological decisions as the key explanations for differences in deforestation estimates.

Remote Sensing of Deforestation - Deforestation in the Brazilian Legal Amazon

The BLA covers approximately 5 million square kilometers, about 61% of Brazil's territory. The BLA also contains 63% of the Amazon biome and is responsible for the largest contribution of deforestation and associated carbon emissions among the countries sharing the Amazon basin. According to INPE (2010), deforestation rates in the BLA increased 40% from 2001 to 2002. INPE data for the period of 2003/2004 indicate deforestation of 27,772 km^2, an area larger than the State of Sergipe. From 2005 deforestation rates show some reduction, going from 19,014 km^2 in 2005 to 6,451 km^2 in 2010 (INPE 2010). Annual deforestation estimates from INPE appear in Figure 2-1. The temporal variation in deforestation suggests changes in the drivers of LULCC over time.

Deforestation in the Amazon has historically followed the construction of roads as well as the spatial expansion of logging and agricultural frontiers (Keller et al. 2009). Roads constitute a key factor inducing the spread of deforestation (Laurance et al. 2001, 2004; Nepstad et al. 2001; Soares-Filho et al. 2004, 2006). In the eastern Amazon, especially along the highway from Brasília to Belém (BR-010) and from Cuiabá to Santarém (BR-163), deforestation during the 1970s and 1980s occurred usually from logging followed by conversion to shifting cultivation and pasture formation (Keller et al. 2009; Serrão and Homma 1993; Serrão and Toledo 1992).

Most forest conversion occurred in an area called the "arc of deforestation" along the rim of the Amazon basin. The arc is a critical area where there is more pressure for deforestation via road accessibility and greater land settlement, resulting in greater LULCC. The westernmost end of the arc of deforestation lies in the eastern portion of the Brazilian state of Acre. There, deforestation has expanded, primarily along the highways BR-364 and BR-317.

Deforestation Drivers and Estimates in the State of Acre, Brazil

Situated in the extreme west of Brazil's North region, Acre borders the states of Amazonas and Rondônia and the countries of Peru and Bolivia. Acre covers 152,581 km^2, or approximately 3% of the BLA and 1.8% of Brazil's national territory (Figure 2-2). Land use in Acre occurs across a range of land tenure types, including agricultural settlements, large cattle ranches, agro-extractive settlements, extractive reserves and other tenure types. Land tenure is another factor that can influence LULCC, insofar as different lands have different rules, as seen in the Amazon (Ankerson and Barnes 2004). About 80% of Acre's population resides in the eastern part of the state. Roughly 31% of Acre's population is rural. According to INPE, as of 2010 88% of Acre was still covered by forest.

Deforestation in Acre follows the same spatial pattern seen in other states of the Brazilian Amazon such as Rondônia and Pará, where most deforestation has happened due to agricultural expansion near road infrastructure (Pfaff et al. 2007; Alves 2002a; Alves 2002b). According to INPE (2010), from 2000 to 2005 Acre exhibited deforestation of approximately 592 km^2 per year. From this period, deforestation is declining from 592 km^2 per year in 2005 to 259 km^2 per year in 2010, a reduction of more the 50%. Deforestation variation in Acre state can be observed in Figure 2-3.

Notably, the two federal highways that cross Acre are experiencing improvements. The BR-317 was completely paved by the end of 2002. Also called the Inter-Oceanic Highway, BR-317 passes through eastern Acre, where deforestation is more prevalent. The analysis will therefore focus on the municipalities in the eastern portion of Acre along the BR-317. The study area therefore encompasses nine municipalities: Assis Brasil, Brasiléia, Epitaciolândia, Xapuri, Capixaba, Senador Guiomard, Plácido de Castro, Rio Branco and Porto Acre. Together these municipalities account for approximately 22% of Acre's territory (Figure 2-4).

Methods of Deforestation Estimation in Acre from Three Sources

Remote Sensing Data Sources

In this chapter, I evaluate three sets of deforestation estimates from different sources with municipal level data for Acre: INPE, IMAZON, and the NSF HSD project at UF. Table 2-1 presents basic information about the remote sensing data from each source. I note that all of the estimates are based on Landsat images with comparable spatial resolutions. However, while the land cover classifications all cover Acre and use recent Landsat data, the exact geographic coverage beyond Acre varies, and the specific dates for which LULCC classifications are available differ.

The INPE data come from the Brazilian Deforestation Satellite Monitoring Project - PRODES (INPE 2010). PRODES data are available for municipalities in Acre for each year from 2000 to 2010. PRODES data are also available for earlier years, but only at the level of Brazilian states. PRODES data cover the entire BLA, including all of Acre.

The IMAZON data come from IMAZON's 2006 report on deforestation in Acre (Souza 2006). These deforestation estimates cover a longer time period, from 1988 to 2004, and are specific to the state of Acre. The IMAZON study was conducted independently of INPE's PRODES program.

The NSF HSD UF project data were acquired independently of INPE and IMAZON's efforts. The UF data cover a period of almost 20 years, from 1986 to 2005. However, the UF data come in 4- or 5-year time steps, unlike the INPE and IMAZON data, which come in 1-year time steps.

Image Processing Protocols

The ability to detect and quantify changes in the Earth's environment in general, and specifically of forest cover, depends on development of clear image processing protocols.

Consistent protocols can help ensure accurate measurement of land cover classes and production of comparable estimates through time for accurate measurement of change. Consistency through time is specifically a challenge, as there is greater potential for similarities and differences among multiple data sets which beg questions about the processing protocols and classification methods behind a series of deforestation estimates.

According to Jensen (2005), there are four fundamental steps in digital image processing of remote sensing data to extract useful information about LULCC: 1) radiometric calibration, 2) geometric correction, 3) mosaicking and 4) classification. INPE, IMAZON, and NSF HSD UF followed somewhat distinct protocols for each of these steps, making each step a potential source of differences in the resulting deforestation estimates. In this section, I will briefly review the four steps used by each data source in order to obtain deforestation estimates and try to see what processing differences may account for differences in the estimates.

Table 2-2 summarizes the first three steps – radiometric calibration, geometric correction and mosaicking for the INPE, IMAZON, and NSF HSD UF data. Radiometric calibration is performed for both thermal and reflective bands to eliminate sources of variability such as noise, differences due to satellite instrumentation, solar elevation angle, solar curve and atmospheric effects (Jensen 2005; Chaves 1996). Radiometric calibration is necessary to ensure comparability among images across both space and time, and generate high quality data without noise or distortions.

Table 2-2 shows that the three sources performed radiometric calibration somewhat differently by following distinct protocols. DGI/INPE is in charge of receiving, processing and distributing images acquired by the Landsat and CBERS (Satélite Sino Brasileiro de Recursos Terrestre) satellites. For radiometric calibration, DGI uses a set of correction coefficients (http://www.dgi.inpe.br/html/radiometria). These coefficients are the same as those used by

Center for the Study of Institutions, Population, and Environmental Change (CIPEC) at Indiana University.

IMAZON applied a protocol for radiometric calibration drawing on an algorithm developed by Carlloto (1999). This algorithm was implemented in ENVI, using interactive computational programming, specifically Interactive Data Language (IDL). The algorithm predicts the values of the bands that are affected by scattering from those that are not, on a pixel-by-pixel basis. It was compared to earlier algorithms for removing spatially-varying haze and was found to better preserve subtle details in the image and spectral balance between bands (Carlloto1999).

The NSF HSD UF project employed a standardized method for radiometric calibration by using the protocol developed by the Center for the Study of Institutions, Population and Environmental Change (CIPEC) at Indiana University (Green1999, 2001). CIPEC uses the same correction coefficients as INPE DGI, but with its own protocol that includes standardized procedures for image registration and calibration, allowing for comparison of results across multiple study sites, a priority of CIPEC research. Calibration was implemented using an Excel spreadsheet. This spreadsheet incorporates specific parameters needed to generate calibration functions in order to convert slope and intercept values from raw DNs to surface reflectance values. The surface reflectance parameter is then incorporated in an ERDAS radiometric model in order to correct the image.Following the discussion above, it is evident that all three sources developed their own radiometric protocols. The issue for the present purpose is whether different radiometric calibration protocols bear important ramifications for deforestation estimates. Bernstein et al. (1983) argue that the effects of the atmosphere upon remotely sensed data do not affect the ability to accurately estimate physical properties of the earth surface, since they are part of the signal received by the sensing device.

The second step, geometric correction, is critical for ensuring that satellite images are correctly located on the surface of the planet and with regard to each other. This is especially important when working with multiple images for a given time point. Geometric correction is also important for cloud removal and when combining satellite imagery with other spatial data sources such as digital elevation models (Jensen 2005).

INPE geometric correction was originally done manually and based on official maps, which led to the propagation of error. From 2000 to 2005, registration was done using recorded images with reference to the previous year. From 2005, INPE has been conducting geometric calibration using orthorectified geocover images released by NASA. The use of orthoimages reduces error propagation since these images are geodetically accurate. Nine GCPs are manually collected and distributed throughout the image, one in each corner of the image, four near the middle of each side and one in the center. INPE only allows for the identification of changes in forest cover areas larger than 6.25 ha. Products are produced at 1: 250,000 scale, which implies georeferencing Root Mean Square Error (RMSE) of up to 125 m. But INPE adopts the criterion of accepting errors of up to three Landsat pixels, or 90m. In the case of Acre uncertainty is critical due to the predominance of small polygons of deforestation in some areas such as extractive reserves and indigenous lands.

IMAZON conducted its geometric corrections manually using an Acre Environment Institute (IMAC) georeferenced image with the year 1999 as a reference image (IMAC 1999). IMAZON used approximately 35 GCPs per image and obtained an RMS lower than 1 Landsat pixel (30 m). This type of image correction is the same used by INPE before 2005 and as such may lead to error propagation. On the other hand, the larger number of GCPs per image collected by IMAZON than INPE, resulted in a smaller RMSE.

The NSF HSD UF team selected reference images for each Landsat image path and row from the Global Land Cover Facility (GLCF) at the University of Maryland (UM). All images were subsequently georeferenced using their respective base image. Similarly to INPE's geometric correction since 2005, the reference image used by NSF HSD UF comes from a GeoCover dataset which was acquired for free and has a higher quality standard. NSF HSD UF used between 45 and 60 GCPs per image distributed as uniformly as possible across each image. The number of GCPs obtained by NSF HSD UF is thus greater than the number collected by INPE and similar to IMAZON. Consequently, NSF HSD UF was able to obtain an RMSE error of 0.5 pixels (15 meters), lower than INPE or IMAZON. Lower RMSE implies a good geometric accuracy which can interfere in the classification accuracy of land cover, since a given location will appear to be in different positions. This may therefore account for differences in deforestation estimates. We can conclude thus, that the geometric protocols developed by the different sources can be one limiting factor that may affect deforestation estimates.

The third step, mosaicking, is necessary when the study area is larger in extent than one satellite image (Jensen 2005). Image mosaicking straddles the overlapping region of two or more images in order to create a single seamless composite image (Jensen 2005). Because eastern Acre spans more than one Landsat image, all three data sources make use of mosaicking.

Mosaicking by INPE is illustrative. Images are mosaicked jut to give an idea of how the images are aligned respective to another image. It does not follow any radiometric protocol for image equalization, that is, no processing was made to smooth out light imbalances, reduce color disparities between images, or remove brightness variations among images. Each image (path/row) was processed, classified individually and manually edited for missing data. IMAZON mosaicks were made using ENVI software, but without many details available for comparison, see (Souza 2006). NSF HSD UF data mosaicking used functions in Erdas Imagine

Mosaicking tool such as Image Dodging, Color Balancing and Histogram Matching. These Erdas functions were used in combination to help smooth out light imbalances, reduce color disparities and remove brightness variations among images. These functions also helped to normalize data among images captured on different days, so that images with slight differences due to sun or atmospheric effects could be normalized, for mosaicking and classification.

Land Cover Classifications

The transformation of spectral data into earth surface information through the extraction of thematic features related to land cover has been traditionally done through classification techniques. There are a large number of classification schemes used for land use and land cover throughout the world. Some present techniques for supervised or unsupervised classification, but they do not always indicate the specific characteristics of each application or endorse a specific protocol for image classification (Jensen 2005; Anderson et al. 2001; Thompson 1996; IBGE 1992; Florida 1999; Gregorio & Jansen 1998; www.landcover.usgs.gov/classes.php; www.africover.org/LCCS).

Of all the steps in satellite image processing, classification is potentially the most important for estimation of specific land cover changes like deforestation. While decisions made in the other steps may indeed result in errors and biases that can affect land cover estimation, land cover classification can have major ramifications. The classes selected, and their relationship to calculating land cover measures, can greatly affect estimates of deforestation and other types of land cover. This is especially the case insofar as there may be different land cover classes used by different sources in calculating deforestation. A key case in point concerns the definition of what constitutes forest cover, and how ambiguous classes are categorized for purposes of deforestation estimation. Secondary growth, or immature forest, may be classified as forest, non-forest, or a third category. Calculations of deforestation are affected in different

classifications that separate immature forest, and if secondary growth is counted as forest or non-forest in deforestation estimation.

Table 2-3 outlines the classification methods used by the three sources and Table 2-4 outlines land-cover classes obtained by each source. INPE used a Linear Spectral Mixture Model (LSMM) and image segmentation before classification. The LSMM estimates the proportion of the components of soil, vegetation and shade by pixel. The LSMM considers the spectral response of each pixel in the various bands of the images, and therefore generates synthetic bands of soil, vegetation and shade. After LSMM, the three synthetic bands produced (vegetation, soil and shade) were resampled to 60 x 60 meters to optimize the digital processing time and minimize disk space used.

Afterward shade and soil fraction images derived from the LSMM are segmented by region growing method, using the thresholds of similarity pre-established through several experiments in work on land use and cover, made in the Amazon. Once segmentation is performed, the synthetic bands produced are used for classification purpose (Câmara et al. 2006). An unsupervised classification algorithm, the Interactive Self-Organizing Data Analysis Technique - (ISOSEG) was used to classify segmented soil and shade synthetic images (Câmara et al. 2006). Images were analyzed in the Information Processing System (SPRING). Land-cover classes were pre-established based on the IBGE vegetation map as forest, non-forest, cloud, water, shadows, and deforestation total and increment (Table 2-4).

Satellite images in the IMAZON data set were first classified using the Interactive Self-Organizing Data Analysis Techniques (ISODATA) algorithm. ISODATA is an unsupervised digital classification method which provides good accuracy at separating classes with different spectral characteristics (i.e., water, soil, forest, pasture). Another advantage is that the ISODATA method allows the user to map areas with complex shapes, like rivers, lakes, and small

deforested areas (IMAZON 2006). ISODATA was implemented with15 spectral classes and a maximum number of classes extracted from the images in a total of 10 iterations, generating the following land cover classes: forest, deforestation, water, cloud, degraded forest and other (beaches, sand banks, ravines and small formations of natural grasslands).

After classification with ISODATA, IMAZON used a visual interpretation method to correct errors in land cover classification. A spatial filter was applied in order to correct errors generated by the automatic classification. IMAZON reclassified forest areas smaller than 0.25 ha to deforestation, since these areas could not be represented in a scale of 1:50.000. IMAZON also applied a temporal filter to ensure that were no errors in classifications among years. This filter is used to detect illogical or impossible transitions in land cover in the time series: for example, an area of deforestation in one year is classified to forest in the second year. Some classification systems would classify that land as under secondary forest, but IMAZON instead reclassified this area as deforestation. After spatial filtering was applied, a 1988 mask was generated and used as reference to map deforestation increments in later years. This procedure was applied for each image pair from 1988 to 2004, generating incremental and overall deforestation maps for each year.

NSF HSD UF first removed clouds, shadows and water from each individual image and from the mosaics before classification using PCA image differencing and thresholding methods (Varlyguin et al. 2001). The generated masks were thus applied to the mosaics in preparation for classification. PCA of satellite images is a statistical process that is widely used to extract useful information from multiple bands by filtering noise from the data.

Before classification UF used also tasseled cap indices (Kauth et al. 1976), mid-infrared index (Boyd and Petitcolin 2004), and 3-by-3 moving window calculation of the variance of each

pixel for bands 4, 5 and 7 for each mosaic generated in order to help as a measure of image texture. Texture is useful for classification of forest versus non-forests (Boyd and Danson 2005).

It was observed that the visible and thermal bands contained striping, limiting the available information for a traditional classification, therefore a rule-based or decision tree classification was applied (Breiman 1984) instead of traditional unsupervised or supervised techniques. The rule based classification approach provided flexibility to eliminate these bands, using only the near- and mid-infrared bands along with secondary derived products.

Field work to support visual interpretation is very important to help classify land cover and improve classification accuracy. Therefore NSF HSD UF conducted field visits for ground truthing in the study region. Field teams collected training samples for accuracy assessment of forest-non-forest (F-NF) classifications in two periods, 2006 and 2007. The 2006 training samples were collected in Acre, Brazil and in Pando, Peru. These points were aggregated to a FNF classification from a finer level of classification and 30 points were selected for each forest and non-forest class. The 2007 training samples were based on a stratified random sample of 300 points within 1.5 kilometers of the Cobija-Sena road corridor for each cover type derived from a preliminary FNF classification using the 2005 mosaic. Of these 300 points, at least ten percent were selected to be visited within the field based on accessibility. Land-cover was then classified into forest, pasture and bare-built, and then pasture and bare-built was aggregated to create the non-forest class.

The three sources differ in terms of bands employed in the classifications methods. The band selection is important to determine the multispectral bands optimal for discriminating one class from another. For image classification, IMAZON and INPE used bands 3, 4, 5 while the NSF HSD UF project used bands 4, 5, 7 (the near and mid-infrared bands) along with secondary derived products. Consequently, UF differs from INPE and IMAZON because UF conducted a

rule-based classification instead of traditional supervised and unsupervised classifications. This technique provided flexibility to eliminate bands with striping, which limit available information from a traditional classification.

Table 2-4 lists the land-cover classes distinguished by each source in order to obtain the forest and non-forest classification. Table 2-5 on the other hand, provides definitions employed by each data source for forest and non-forest classification. INPE defined forest classes using Brazil's official technical manual of vegetation (IBGE 1992). Brazil's Institute of Geography and Statistics (IBGE) has its own vegetation classification scheme that can help classification of land cover data obtained by remote sensing. IBGE's vegetation manual distinguishes different types of forest according to type of vegetation cover. Vegetation types distinguish different forms of vegetation, such as forest trees and shrubs (Cerrado), grassy-woody (Cerrado with Clear Field), etc. This reliance on the IBGE classification differentiates INPE's classification from IMAZON and NSF HSD at UF. INPE also classified water clouds and shadows. Deforestation is generated considering previously defined classes. Where classification is based on the class attributes statistical region within certain acceptance thresholds predetermined equal to 95% or 90%, depending on the complexity of the landscape investigated (INPE 2006).

IMAZON, does not remark how they classify forest cover. Beside forest they also classify clouds, shadows, degraded forest, deforestation (total and increment), beaches, sand banks, and ravine sand small formations of natural grasslands.

Land-cover classes for the UF NSF project were defined in order to evaluate the impacts of road paving and other forms of infrastructure construction and upgrades on forest cover. The UF NSF project therefore considers forest and non-forest classes. The non-forest class stated in Table 2-5 includes pasture, bare fields and urban built land cover, which were all classified as non-forest. Forest cover includes all dense vegetation cover, which includes secondary

succession (generally 3-5 years of age in the study region). Although Table 2-5 does not show water, clouds and shadows classes for UF NSF project, it is important to point out that these classes were removed from each individual image and from the mosaics before classification. Hence while the UF classification does not include water, cloud or cloud shadows, that is because UF masked out those covers prior to classification.

The different classification schemes (Table 2.4) and definitions (Table 2.5) adopted by each source may be responsible for differences in deforestation estimates. It is therefore important to follow an established classification system instead of developing new schemes that may only be used by the producer. According to Jensen (2005), adoption of an existing broadly recognized classification system allows comparisons of the significance of classifications produced by different sources. LULC classes should therefore be selected in order to allow valid comparisons among data sources. This requires a classification system containing consistent definitions of LULC classes among sources.

The classifications system in the three sources presented here is an example of a need for a standardized classification system that has to contain a consistent definition for LULC classes to permit a valid comparison of estimates. Tables 2.1, 2.2, 2.3, 2.4 and 2.5 show several differences among data sources particularly decisions at each of the three steps in image processing, and definitions of land-cover classes as they influence classification methods. Table 2.6 shows a summary of points that could be a reason to expect higher or lower deforestation estimates from one source or another.

Among the items listed in Table 2.6, I highlight four that might be especially important explanations for different deforestation estimates between sources. The first two concern the definitions of deforestation and secondary forest adopted by each source. Issues of forest and non-

forest definition and classification class, constitute important factors to explain the dataset discrepancies.

INPE considers deforestation to be anthropogenic modifications in mature forest for development of agriculture and cattle pasture which may give a lower deforestation estimate since only modification of mature forest is incorporated into the deforestation estimate; on the other hand, INPE consider forest regrowth or areas in process of secondary succession as deforested areas which put INPEs estimation higher when compared to other sources. Similarly, IMAZON considers secondary forest as a deforested area. IMAZON also considers as deforestation, all forest areas smaller than 0.25ha, which may be one of the reasons for IMAZON deforestation estimates to be slightly similar to INPE estimates. Different from INPE, UF NSF considers deforestation to include anthropogenic activity, all pasture areas and bare/built soil, not only areas resulting from primary forest. UF NSF also considers secondary succession as forest, since dense canopy is achieved within 3-5 years within this region. Consequently, to the extent that secondary vegetation covers eastern Acre, UF deforestation estimates may be relatively low compared to both INPE and IMAZON estimates.

The third point to consider regards classification decision concerning to clouds. INPE estimated areas deforested under clouds, while the other sources did not. Specifically, INPE assumes that the proportion of cleared areas under clouds is the same as the observable areas. By contrast, IMAZON classified clouds, shadows and water as an independent class, and made no assumption about deforestation in this class. Further, the UF NSF project removed clouds, shadows and water from each image before classification. It is possible but unclear how these differences will affect deforestation estimates by INPE as opposed to IMAZON and UF. If deforestation under clouds is higher than elsewhere, INPE estimates will underestimate deforestation; if deforestation under clouds is less than elsewhere, INPE estimates will overstate

deforestation. In addition, there remain questions of the extent of cloud over, which may vary among images, even for the same path, row and year, if images come from different dates.

The last point that could be the reason to expect higher or lower deforestation estimates from one source to another is the scale. Issues of scale can be relevant to explain differences between the two datasets, since coarse representation might reduce estimates of deforestation by leaving out small clearings. The coarse resolution employed by INPE, for example, leds both to underestimation of deforested areas in cases where forest clearing occurs in small plots, and to overestimation of deforestation in landscapes with small forest patches.

Comparative Analysis of Deforestation Estimates in Acre

Given the foregoing discussion of similarities and differences in the data sources and processing protocols, it is possible that deforestation estimates will vary among the data sources for eastern Acre. This section therefore offers a comparative analysis of deforestation for eastern Acre - Brazil, using estimates from INPE, IMAZON, and UF.

Direct comparisons require comparable data for geographic areas and time periods. The comparisons however pose challenges in that the three sources do not have the same protocols. The comparable geographic coverage in eastern Acre nonetheless opens analytical possibilities to identify the most important explanations for any differences in deforestation estimates observed. Specifically, the analysis that follows allows an evaluation of the importance of different steps in processing protocols and classification systems in order to account for differences in deforestation estimates. As a result, we gain insights into the reasons for differences in deforestation estimates, and thus an idea as to what elements of deforestation estimation may require the greatest caution when producing new estimates.

Figures 2-5, 2-6, and 2-7 present cumulative percentages of areas deforested in the nine municipalities in eastern Acre from INPE, IMAZON, and the UF NSF project, respectively. All

three figures show varying levels of deforestation among the municipalities. The municipalities with relatively high deforestation in one source are the same as the "high deforestation" municipalities in the other sources. There are four more or less distinct groups of municipalities in terms of their relative deforestation percentages. The first group comprises the municipalities that had lost most (50-70%) of their forest cover, such as Senador Guiomard and Plácido de Castro. The second group includes Capixaba, Epitaciolândia and Porto Acre, with 30-50% deforestation. The third group includes Rio Branco, Xapuri and Brasiléia with 20-30% forest loss, and the fourth group contains only one municipality, Assis Brasil, with under 10% deforestation. What is more, all three figures indicate increasing deforestation over time, especially in the first group. Thus in terms of variation among municipalities and dynamics over time, the three sources broadly exhibit similar findings for deforestation percentages. In these respects, the estimates in these figures are similar.

That said, more direct comparisons for specific time point and municipalities among the data sources reveal differences in deforestation estimates. The remainder of this section focuses on such direct comparisons. Specifically, the remainder of the analysis proceeds in two steps, First, I observe differences in the time points available from the three data sources, in order to identify common time points to permit direct comparisons of deforestation estimates among the sources. And second, I focus on those common time points and make comparisons in order to calculate quantitative differences in deforestation and to observe which have higher and lower deforestation estimates.

With regards to the time period for which deforestation estimates are available, INPE has the longest time series of satellite data for the BLA, dating back to 1978. INPE deforestation estimates for Brazilian municipalities in the BLA run annually from 2000 to 2009. By contrast,

the IMAZON data includes annual deforestation estimates from 1994 to 2004 for municipalities in Acre. The UF NSF project data refer to 1986, 1991, 1996, 2000, and 2005.

The different time periods and time steps for which deforestation estimates are available present limitations to the number of direct comparisons permitted by the three data sources. All three sources have deforestation estimates for the year 2000. I also interpolated the UF data between 2000 and 2005 to obtain an estimate for deforestation in 2004 in order to secure a second time point for comparisons. I therefore focus on deforestation estimates for the years 2000 and 2004, shown in Tables 2-7 and 2-8, respectively.

Comparisons of deforestation percentages for the municipalities in eastern Acre reveal differences in deforestation estimates. For example, whereas INPE and IMAZON both indicate that deforestation in the municipality of Senador Guiomard in 2000 was roughly 55%, the UF NSF project indicates a value of around 33%. Further, by 2004/2005, INPE data suggest a higher deforestation percentage in Plácido de Castro than Senador Guiomard, whereas IMAZON data suggest comparable percentages and UF indicates higher percentage in Senador Guiomard than Plácido de Castro.

In general, by the estimates of deforestation presented in Tables 2-7 and 2-8, we can conclude that UF deforestation estimates are substantially lower than those presented for INPE and IMAMAZON. INPE and IMAZON estimates for some municipalities differ by almost 50% when compared to UF NSF. Although differences between INPE and IMAZON are usually higher than NSF differences among INPE and IMAZON are lower. An inquiring observation is observed; where deforestation estimates from INPE which was usually higher than IMAZON in 2000, in 2005 it presented inverse results, that is INPE estimates was lower than IMAZON.

In summary, data presented here show that INPE and IMAZON “overestimate” the deforestation estimate and UF NSF project “underestimate” it. This difference in deforestation

estimates is associated to image processing protocols, land-cover class definition, and spatial scale, which in some cases, when associated to definitions of deforestation employed as part of the different analysis of deforestation.

A clear image processing protocol can help ensure accurate measurement of land cover classes and production of comparable deforestation estimates. INPE, IMAZON and UF NSF as described in this paper, followed somewhat distinct protocols for radiometric calibration, geometric correction, mosaicking and classification, which make each step a potential source of differences in the resulting deforestation estimates.

Radiometric calibration is necessary to ensure comparability among images across both space and time, and to generate high quality data without noise or distortion. But it is possible that differences in radiometric calibration could be not one of the main factors responsible for differences in estimates between sources. This hypothesis can be confirmed, since INPE and IMAZON follow different processing protocols for radiometric calibration, however, both sources present similar deforestation estimates. Likewise, INPE and UF NSF use the same algorithm developed by Chander et al. (2009) for radiometric calibration, however deforestation estimates among them are highly different.

Geometric correction is critical for ensuring that satellite images are correctly located on the surface of the planet and with regard to each other. We can state that geometric correction is important for image processing, but not a crucial factor responsible for differences in estimates between sources. This hypothesis can be confirmed, since IMAZON follows the same image correction used by INPE before 2005. IMAZON also used a number of GPSs per image higher than INPE, resulting in a smaller RMS then INPE; however IMAZON has actually comparable estimates as INPE. UF NSF, also similarly to INPE's geometric correction since 2005 used reference image from a Geo Cover dataset (high quality standard dataset), and used a number of

GPSs per image higher than INPE and IMAZON, resulting in a smaller RMS showed lower deforestation estimates.

Mosaicking is important when the study area is larger in extent than one satellite image. It straddles the overlapping regions of two or more images in order to create a single seamless composite image. INPE, different from the UF NSF project, does not apply any processing techniques to smooth out light imbalance, reduce color disparities and remove brightness variation among images. Mosaicking for INPE was just illustrative; images were mosaicked jut to give an idea of how images are aligned respective to another. Again, mosaicking processing is important but not crucial for deforestation estimate, since UF NSF project also followed a standardized process for mosaicking when compared to INPE, however their deforestation estimates were highly different. Comparison could not be made with IMAZON since the source did not provide detailed steps for image mosaicking.

All image processing steps, like radiometric calibration, geometric correction and mosaicking may indeed result in errors and biases that can affect land-cover estimates. Differences in definitions and land-cover classification schemes adopted by each source are potentially the most important factors for estimation of specific land-cover changes like deforestation. The way that each source selected its land-cover classes, and their relationship to the calculation of land cover measures, can greatly affect estimates of deforestation and other types of land cover, this can be confirmed by analyzing the different land cover classes used by the different sources in calculating deforestation estimates.

For example, INPE classified forest cover using IBGE vegetation maps, where different types of forest are distinguished. UF NSF classified as forest all dense vegetated cover, which includes secondary secession, since dense canopy in Acre regions is achieved within 3-5 years of vegetation regrowth. On the other hand, INPE and IMAZON, different from UF-NSF, classified

forest regrowth or areas in the process of secondary succession as deforested areas. The way secondary vegetation is handled by each source is a key point to answer the question of why estimates among sources are different.

Definition of what constitutes forest cover, what constitutes deforestation, and how ambiguous classes are categorized for purposes of deforestation estimation is critical. Secondary growth, or immature forest, may be classified as forest, non-forest, or a third category. Calculations of deforestation are affected in different classifications as separate immature forest, and whether secondary growth is counted as forest or non-forest in deforestation estimation.

Spatial scale, when associated to definitions of deforestation employed as part of the different analysis of deforestation, may be one fact that affects estimate differences. For example, INPE covers a geographic area of 500 million ha, the entire BLA, but analyses focus on the remote sensing platform, usually Landsat, which has a spatial resolution of 30 x 30 meters, covering an area of 900 m^2. This area is afterward resampled to 60 x 60 meters. IMAZON, on the other hand, covers a small geographic area (153,149.9 km^2), the entire state of Acre. Although IMAZON uses the same remote sensing platform and spatial resolution for their deforestation analysis as INPE, deforestation definitions employed in the analysis consider areas smaller than 2,500 m^2 as deforestation, which are more than 2 pixels. This may be one of the reasons why IMAZON deforestation estimates in some cases are higher than INPE estimates.

The UF NSF project analysis covered an area of approximately 300,000 km^2, the region of Madre de Dios (Peru), Acre (Brazil) and Pando (Bolivia) - MAP region. Like INPE and IMAZON, the remote sensing data is from Landsat and spatial resolution of 30 x 30 meters, representing a pixel area of 900 m^2. Its forest definition considers all secondary successions as forest (Table 2.6). Coarser resolution analysis therefore, can underestimate deforestation taking place in small plots, as well as overestimate deforestation in areas with remaining small patches

of forest. But the issue of definition, independent of size of geographic area and spatial resolution, has decisive influence on the classification outcomes and deforestation estimates. Hence, whether a source under or overestimates deforestation is a relative issue linked to its LULC definition, in this case how they define forest, non-forest and secondary forest.

Conclusion

From the analysis, it is possible to say that the image processing steps followed by each source like radiometric calibration, geometric correction and mosaicking are very important to obtain data accuracy; however it is not crucial to avoid differences among deforestation estimations among sources.

Differences in definitions and land-cover classification schemes adopted by each source are potentially the most important factors for estimation of specific land-cover changes like deforestation. The way that each source selected its land-cover classes and their relationships to calculating land-cover measures can greatly affect estimates of deforestation and other types of land cover. This can be confirmed by looking at the differences in land-cover classes used by the different sources in calculating deforestation estimates.

Definition of what constitutes forest cover, what constitutes deforestation and how ambiguous classes are categorized for the purpose of deforestation estimation are also crucial. INPE, IMAZON and UF NSF classify secondary growth in different ways. For example, for different sources immature forest may be classified as forest, non-forest, or a third category. Therefore calculations of what is forest and what is deforestation are affected by different classification schemes and definitions adopted by each sources resulting therefore in different estimates among sources. INPE and IMAZON separate immature forest and secondary growth from forest cover, and they consider secondary growth as deforestation; on the other hand, UF NSF includes secondary growth as forest.

How secondary growth is categorized has a huge impact on deforestation estimates. Deforestation estimates have been an important metric to evaluate the effectiveness of environmental policies, particularly programs involving payment for environmental services (PES). This raises questions by the deforestation estimates users about which available deforestation definition is best for what purpose, and if deforestation definition is a sufficient concept itself for land-cover monitoring applications, especially for PES programs.

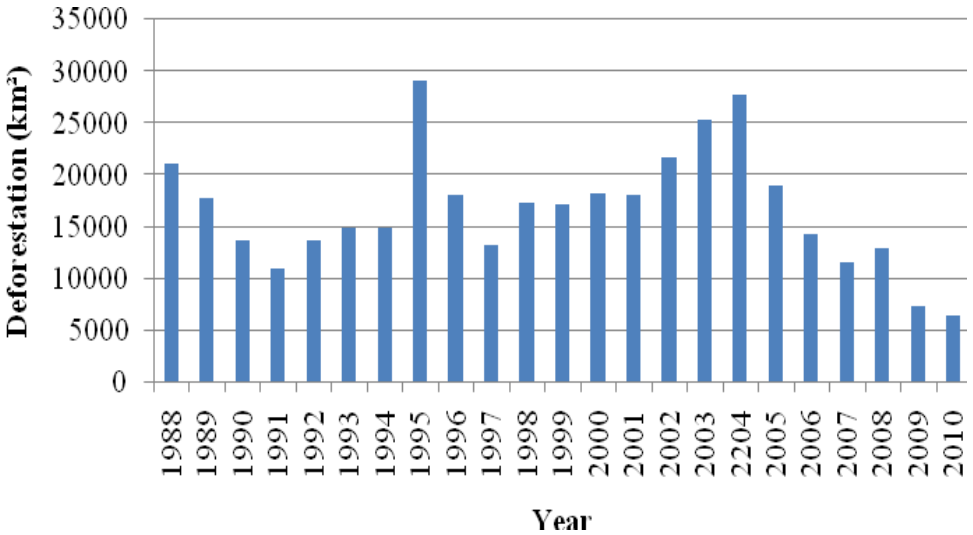

Figure 2-1. Deforestation in the Brazilian Legal Amazon (1988 to 2010). Source: INPE (2010).

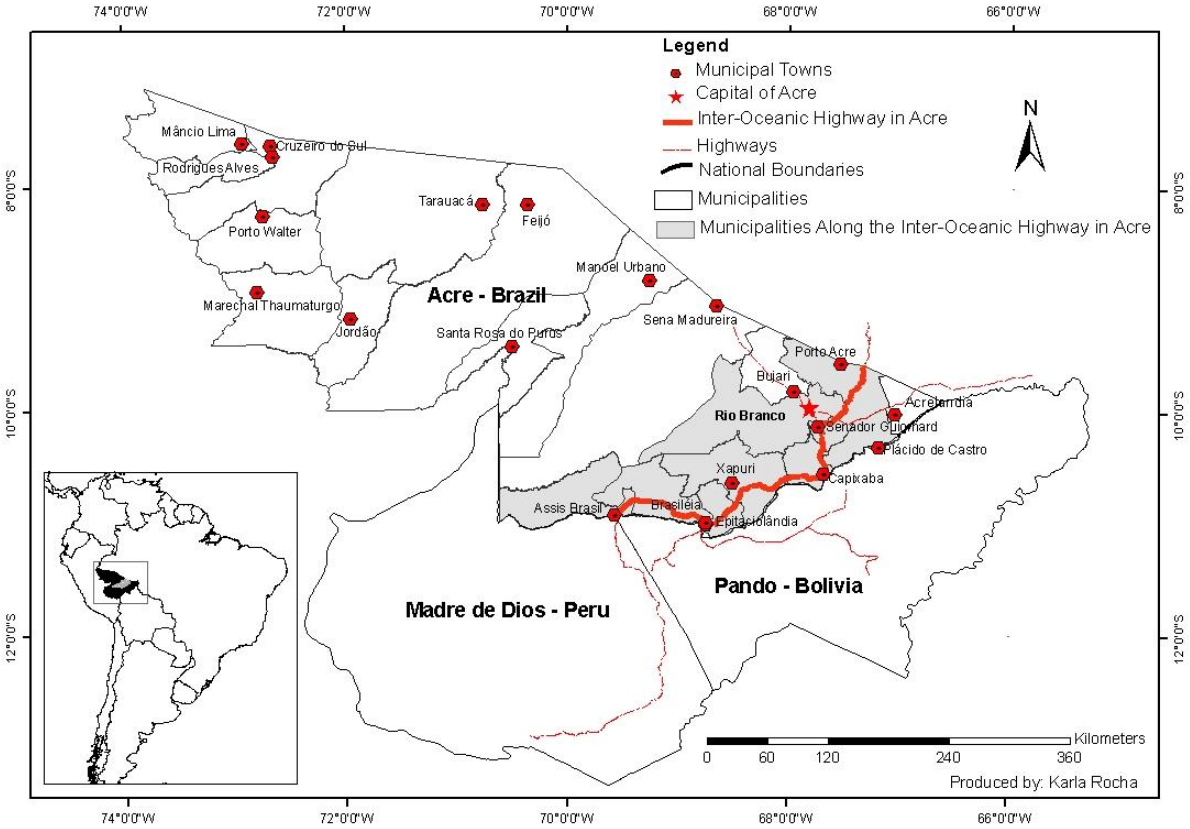

Figure 2-2. Acre State.

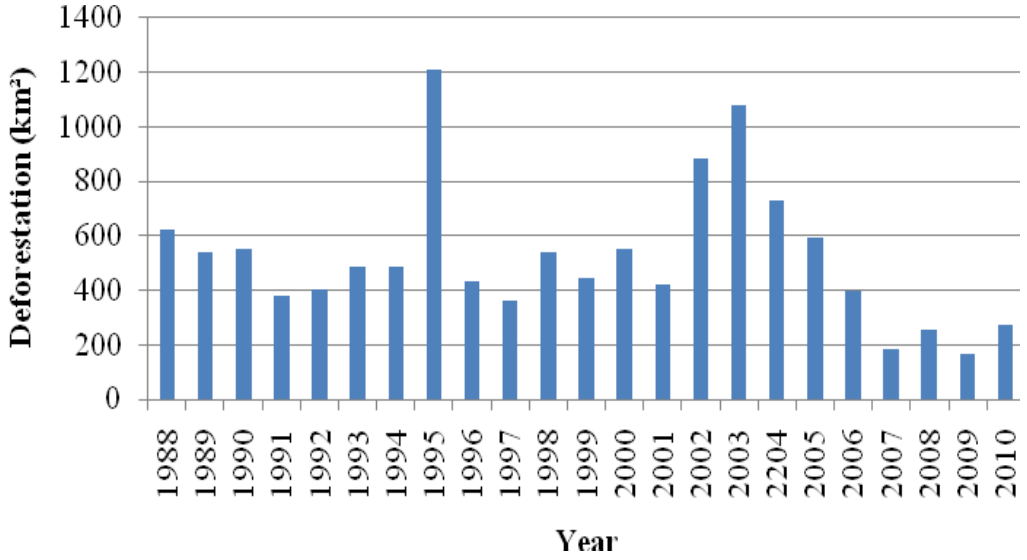

Figure 2-3. Deforestation dynamics in Acre (1988-2010). Source: INPE (2010).

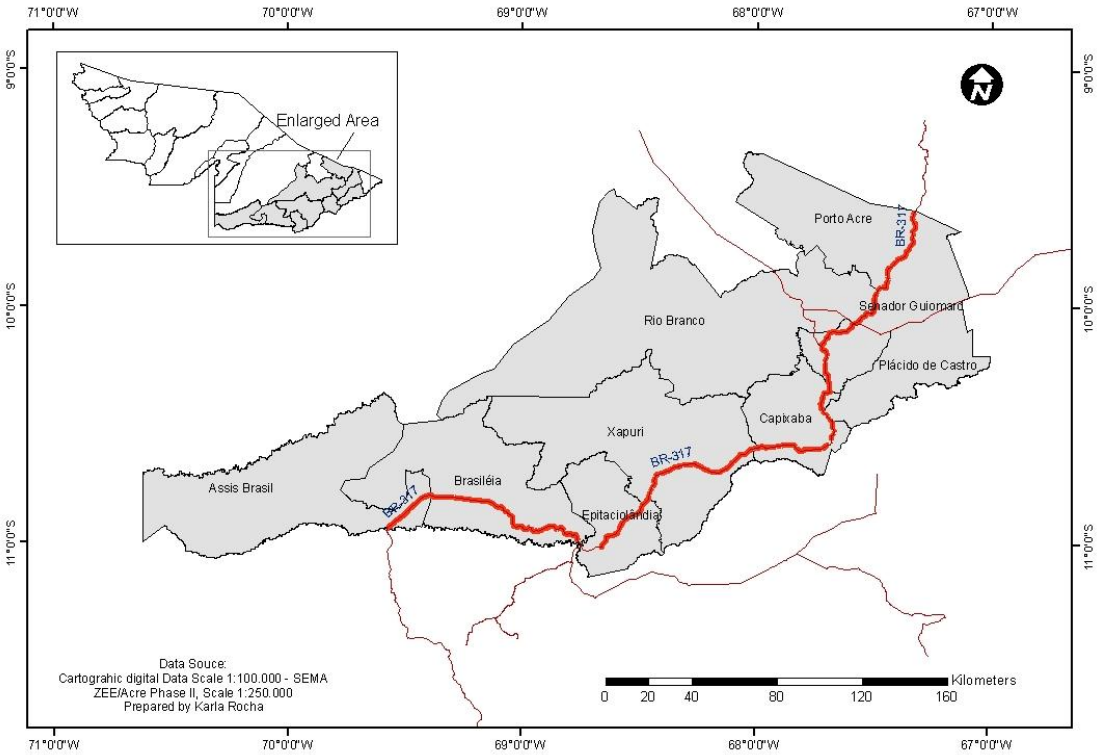

Figure 2-4. Study Area.

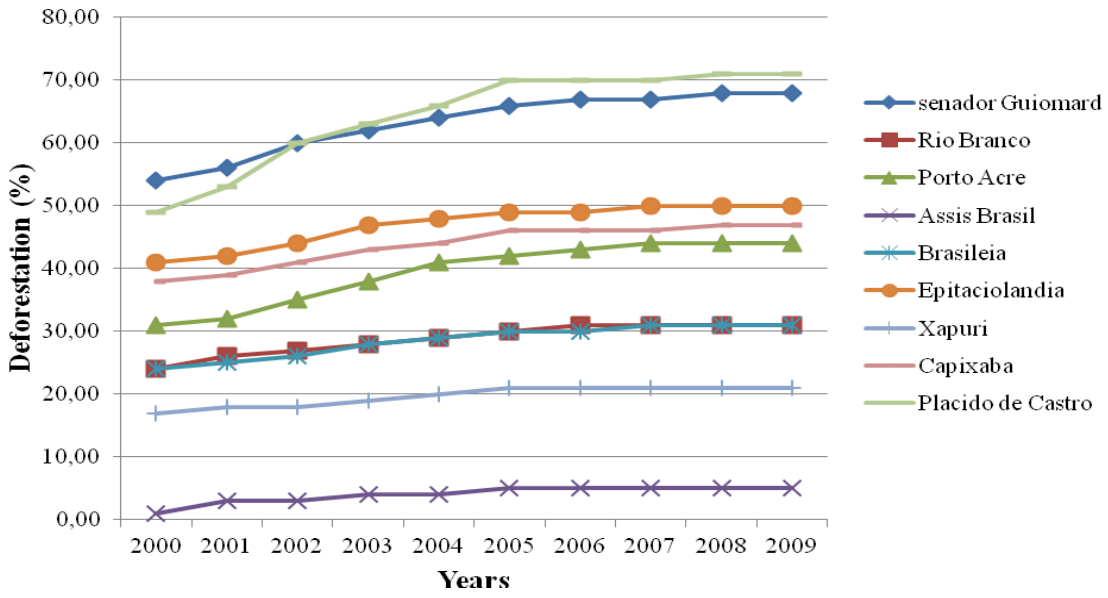

Figure 2-5. Deforestation percentages in municipalities of eastern Acre, Brazil from INPE data for 2000 to 2009.

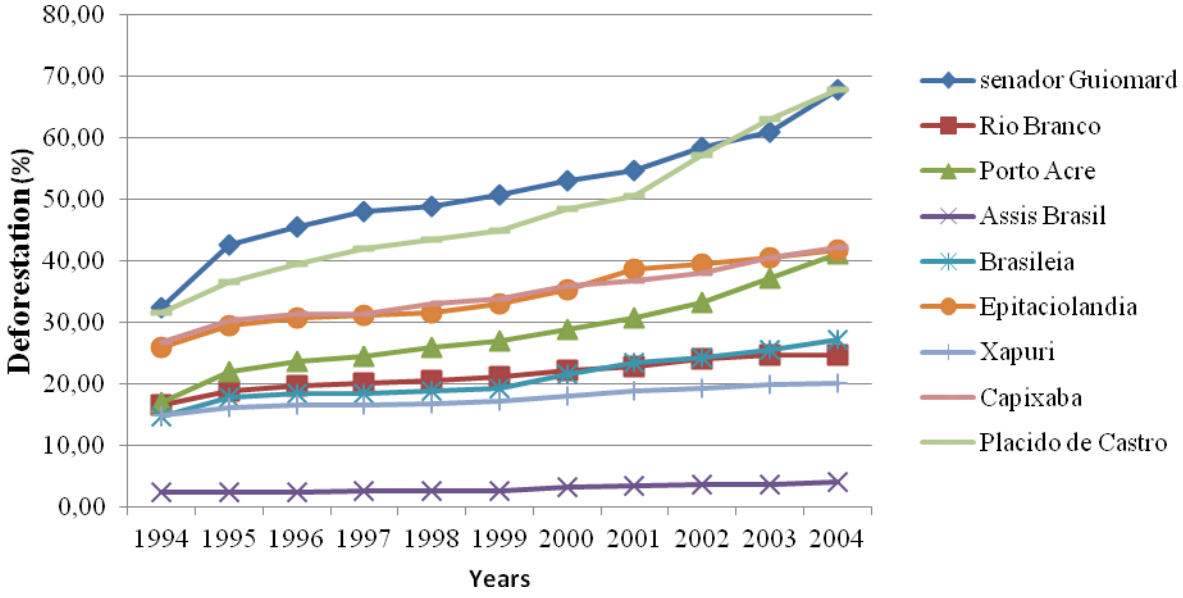

Figure 2-6. Deforestation percentages in municipalities of eastern Acre, Brazil from IMAZON data for 1994 to 2004.

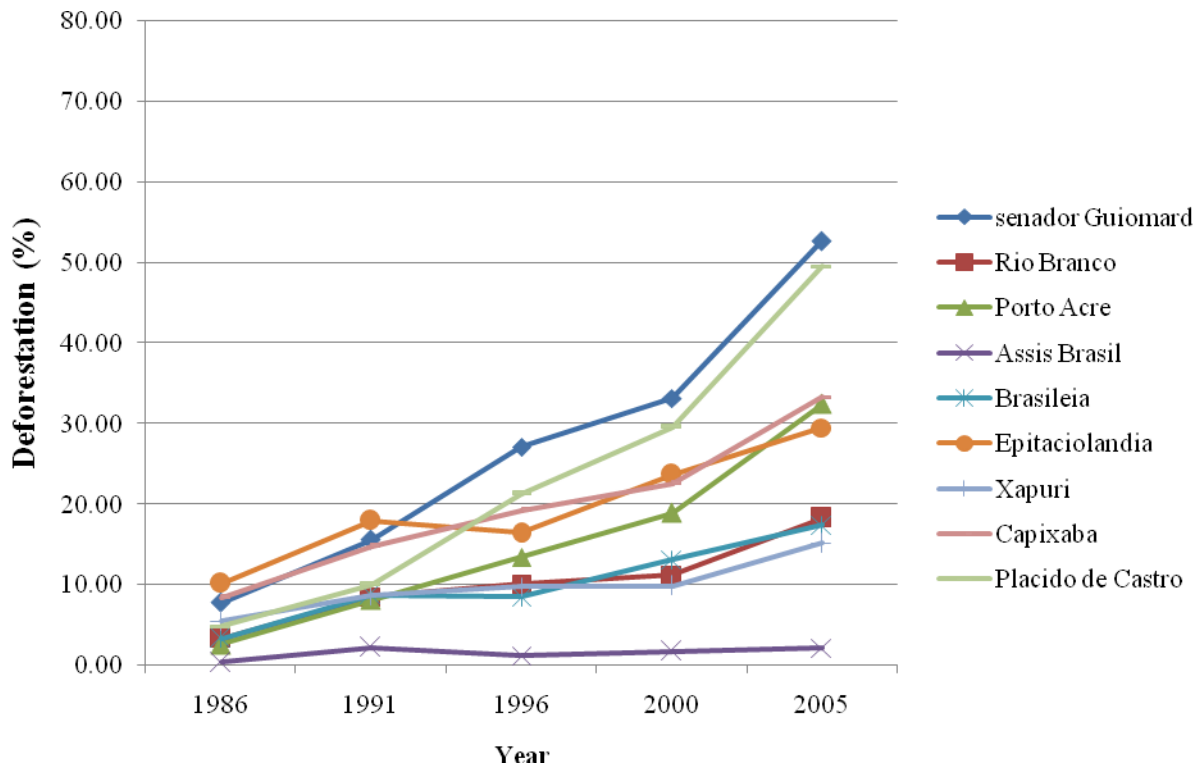

Figure 2-7. Deforestation percentages in municipalities of eastern Acre, Brazil from NSF HSD UF data from1986 to 2005.

Table 2-1.Comparison of Remote Sensing data sources for municipal estimates in Acre, Brazil.

Features	Data set INPE	IMAZON	NSF HSD UF
Satellite	Landsat TM	Landsat TM	Landsat TM and ETM+
Spatial Resolution	Originally 30 x 30 m, then converted to 60 x 60 for the final data	30 x 30 m	30 x 30 m
Time Period	2000-2009	1988-2004	1986-2005
Geographic Coverage	Whole Brazilian Legal Amazon – 5,217.423 km²	Entire Acre state – 153,149.9 km^2	MAP Region – 300,000 km^2

Table 2-2. Digital images processing of Remote Sensing data for three sources of deforestation estimates for Acre, Brazil: Calibration, Geometric Correction, and Mosaicking.

Basic processing steps	PRODES/INPE	IMAZON	NSF HSD/UF
1. Radiometric Calibration	Color composite images are obtained already corrected by DGI, which is in charge of receiving, processing and distributing LANDSAT and CBERS data. For radiometric calibration, DGI uses algorithms developed by Chander et al. 2009.	Algorithm developed by Carlloto (1999) implemented using EVI 4.2 software and Interactive Data Language (IDL).	Standardized method with protocol developed by CIPEC (Green et al. 1999; Green et al. 2001). Protocol implemented using ERDAS modeling and **a**lgorithm developed by Chander (2003).
2. Geometric correction	From 1997, made manually and based on official maps, which led to error propagation. Later images were registered from image to image and from 2005, using orthorectified images released by NASA	Image to image using IMAC georeferenced image year 1999 as reference image.	Image to image using University of Maryland Global Land Cover - Geocover 2000 images;
2.1- Number of Ground Control Points (GCPs) per Image	Usually 9 GCPs	35 GCPs	40 to 60 GCPs
2.2- Root Mean Square Error (RMSE)	2 pixels (90m)	Less than 1 pixel (30m)	Less than 0.5 pixel (15m)
2.3- Correction Algorithm	Polynomial Algorithm	Polynomial Algorithm	Polynomial Algorithm
2.4- Software	Spring	ENVI 4.2.	ERDAS 9.3.
3. Mosaicking	Made in SPRING, after classification	ENVI Software	Made using Erdas Software Mosaicking tool: Image Dodging, Color Balancing and Histogram Matching

Table 2-3. Land-cover classification methods for estimating deforestation in Acre, Brazil employed by INPE, IMAZON, and NSF HSD UF.

	INPE	IMAZON	NSF HSD UF
1. Bands Used	- Bands used 3, 4, 5	- Bands used 3,4, 5	- Bands used 4,5,7 (due to striping in visible and thermal bands)
2. Pre-Classification	- Linear Spectral mixture (LSM) - Segmentation - Resampling to 60 x 60 m. - Generation of synthetic bands for soil, vegetation and shade	- No processing pre-classification	- Clouds, shadows and water were removed using a PCA image differencing and thresholding method - Masks were generated and applied to the mosaics in preparation for classification. - Tasseled cap indices, mid-infrared index and a 3by 3 moving window calculation of the variance of each pixel for bands 4, 5 and 7 (as a measure of image texture) were generated for each mosaic
3. Classification methods	- Unsupervised classification-ISOSEG algorithm	- Unsupervised Classification with ISODATA algorithm in ENVI 4.2 - Use of Spatial filters and Visual Interpretation Class Edit in ENVI 4.2 - Annual increment of deforestation were applied for each coming pair of image - Temporal filter - Visual Interpretation (using field data) if needed	- Rule-based or decision tree classification using data mining software Compumine -Field data and visual interpretation used to create decision rules - Result classified image were subset to the East Acre state.

Table 2-4. Land-cover classes from INPE, IMAZON, and NSF HSD UF.

Mapping classes	Data set INPE	IMAZON	NSF HSD UF
Forest	X	X	X
No forest	X	X	X
Water	X	X	
Clouds	X	X	
Shadows		X	
Degraded forest		X	
Deforestation (total and increment)	X	X	
Beaches, sand banks, ravine sand small formations of natural grasslands		X	
Pasture			X
Bare fields and built			X
No data			X
Classification Accuracy	-	95%	93%

Table 2-5. Definitions of land cover classes: INPE, IMAZON, and NSF HSD UF project.

	Sources		
Classes	INPE	IMAZON	NSF HSD UF project
Forest	Areas identified as forest in the IBGE vegetation map (IBGE, 1992)	Not defined by the source	All dense vegetated cover. Includes secondary succession since dense canopy is achieved within 3–5 years of vegetation regrowth in Acre
Deforestation/ non forest	Deforestation is the conversion of areas of primary forest by anthropogenic activity for the development of agriculture and cattle raising, detected from orbital platforms (INPE 2000). Areas in the process of secondary growth after clearing are considered as deforested.	Forest areas smaller then 0.25ha were classified as deforestation due to scale representation 1:50,000.	Pasture areas, and bare/built areas aggregated and classified as non-forest

Table 2-6. Summary of points that could be the reason to expect higher or lower deforestation estimates from one source or another.

Methodological decision	Implications for deforestation estimates by source		
	INPE	IMAZON	NSF
1. Deforestation definition	Deforestation is the conversion of areas of primary forest by anthropogenic activity for the development of agriculture and cattle raising, detected from orbital platforms	Does not make clear how they classify deforestation but they includes all forest areas smaller then 0.25ha or 2.500 m2 as deforested	Deforestation is anthropogenic activity - Pasture areas, and bare/build were aggregated and classified as non-forest.
2. Classification of secondary forest	Forest regrowth or areas in the process of secondary succession are considered as deforested.	Secondary successions are considered as deforested	Secondary successions are considerate Forest. Since dense canopy is archived within 3–5 years within in this region
3. Classification of Clouds	Estimate areas deforested under clouds. The estimate assumes that the proportion of cleared areas under the clouds is the same as the areas observed as forest in the image.	Clouds, shadows and water were classified independently	Removed clouds, shadows and water from each image before classification
4. Scale representation	Cover entire- BLA - 500 million ha	Cover all Acre state - 153 149, 9 km2	Cover the MAP Region - 300,000 km^2

Table 2-7. Deforestation estimates from each source (2000).

	INPE	IMAZON 2000 (km2)	NSF	INPE	IMAZON 2000 %)	NSF
AssisBrasil	35.60	174.19	88.65	1.00	3.30	1.78
Brasiléia	1,096.50	871.28	513.36	24.00	21.70	13.12
Capixaba	662.90	640.54	384.93	38.00	35.90	22.62
Epitaciolandia	708.70	581.09	391.87	41.00	35.30	23.70
Placido de Castro	1,038.70	1,025.92	575.67	49.00	48.40	29.63
Porto Acre	943.90	831.26	493.83	31.00	28.90	18.93
Rio Branco	2,321.80	1,825.41	992.59	24.00	22.20	11.24
SenadorGuiomard	1,013.60	1,343.54	769.93	54.00	53.00	33.18
Xapuri	920.20	917.32	526.59	17.00	18.00	9.85

Table 2-8. Deforestation estimates from each source (2005).

	INPE	IMAZON 2005(km2)	NSF	INPE	IMAZON 2005 %)	NSF
Assis Brasil	138.2	251.0471	108.59	5.00	5.04	2.18
Brasiléia	1360.4	1240.396	681.89	30.00	31.66	17.43
Capixaba	805.1	812.5039	568.03	46.00	47.89	33.37
Epitaciolandia	840	780.9699	488.22	49.00	47.19	29.53
Placido de Castro	1458.9	1482.962	962.44	70.00	76.23	49.54
Porto Acre	1299.3	1223.322	845.87	42.00	46.89	32.42
Rio Branco	2860.8	2394.602	1616.93	30.00	27.11	18.31
Senador Guiomard	1251.3	1708.637	1222.98	66.00	73.63	52.71
Xapuri	1115.6	1214.272	810.73	21.00	22.71	15.17

CHAPTER 3
LAND TENURE, ROAD, AND DEFORESTATION PATTERNS IN SOUTHEAST STATE OF ACRE – BRAZIL

Summary

This paper analyzes land tenure, road, and deforestation linkages in the southeastern part of Acre-Brazil. This region has been integrated by the Inter-Oceanic Highway, which has recently been paved in Acre and will link Brazil to markets in the Pacific via Peru. Highway paving will facilitate the extraction and transport of forest products and increase cattle ranching through improved access to formerly remote areas. However, land tenure arrangements are diverse in eastern Acre, so it is likely that deforestation will vary among different lands along the Inter-Oceanic Highway. This chapter therefore evaluates deforestation over time among lands with different use rules along the Inter-Oceanic Highway in eastern Acre. I employ a time series analysis of Landsat Thematic Mapper (TM) imagery from 1986, 1991, 1996, 2000, and 2005 to evaluate the spatial and temporal distribution of deforestation. The analysis highlights the interaction of highway paving through time with distance from the highway, and diverse land and tenure rules. Results show that deforestation estimates of the selected lands by time of completed paving status have accelerated after highway paving; almost all segments show an exponential increase in deforestation estimates. In addition, all three explanatory factors – time since paving, distance from highway, and land tenure – are important for understanding where deforestation is greater. However, the acceleration in deforestation is a generalized phenomenon, and occurs in all lands considered, regardless of their time since paving, distance to highway, and land tenure type. This acceleration is cause for concern, because recent deforestation estimates are nearing legal limits in some tenure types and have exceeded limits in others. This raises policy questions concerning land use rules alongside initiatives for regional development via highway paving.

Introduction

Tropical deforestation is considered to be one of the most significant types of land-cover changes underway globally (Myers 2000, Lambin and Geist 2006). It also has a substantial effect on climate change through burning and release of CO_2 into the atmosphere (Fearnside 2008). Deforestation and climate change are expected to result in considerable water shortages and other forms of resource scarcity (Rayner and Malone 2001). These concerns raise questions about the spatial distribution of deforestation, which influences the locations where many negative environmental consequences are likely to result. Information about the spatial distribution of deforestation is necessary to estimate the impacts of habitat destruction and fragmentation on biological diversity (Skole and Tucker 1993, Laurance et al. 1997, 2004).

No region of tropical deforestation has drawn more concern than the Amazon basin, especially the Brazilian Amazon (Keller et al. 2009). The Amazon forest plays an important role in both moderating temperatures and in recycling water into the atmosphere during the dry season, on local and regional levels. Consequently, large-scale deforestation in the Amazon may result in warmer and drier conditions in the region (Foley et al. 2005; Malhi et al. 2008).

Deforestation in Amazonia has historically followed the extension of roads (Perz et al. 2008) and the consequent expansion of logging, cattle ranchers and agricultural frontiers (Wood 2002). Roads facilitate access to natural resources, raising land values and attracting population and investment. In forested regions, improvements in access may lead to the onset or acceleration of deforestation over time. Further, in spatial terms, deforestation primarily proceeds on the most accessible lands, and then follows a diffusion process over time (Southworth et al. 2011).

That said, simple models of accessibility incorrectly predict deforestation insofar as land tenure rules vary from one place to another. Theoretical frameworks to explain land cover

change highlight the importance of institutional factors, notably land tenure rules, since institutions define use rules that in turn can determine whether and where deforestation and other land use may occur (Geist and Lambin 2006). Hence lands with similar accessibility but different tenure rules may nonetheless exhibit very different deforestation rates and patterns.

In the Amazon, highway paving as well as tenure diversification have proceeded apace. It is therefore likely that any accounting for deforestation in the Amazon requires attention to both factors. In this study, I consider the importance of highway paving, distance from highway and land tenure for deforestation in eastern Acre, Brazil. Acre is a very useful study case because it has incurred high-profile infrastructure investments as well as serving as a policy laboratory for several innovative land tenure models, including some which have since diffused to other parts of Brazil. The analysis focuses on comparisons in deforestation over time and across space in eastern Acre using a time series data set of Landsat images from 1986 to 2005. The imagery span the period of highway paving in eastern Acre, and an area that encompasses locations close to and far from a key highway, along with numerous lands with different tenure rules.

Highway, Conflict and Land Tenure Change in Acre

According to Cavalcanti (1994), land occupation in the state of Acre was facilitated by the state road network, which includes two highways that crisscross the state and thus served as corridors providing access to large swaths of land permitting land settlement and agricultural expansion. With the opening of highways into the Brazilian Amazon in the 1960s and 1970s, and government-supported colonization programs in the 1970s, the political economy of Acre experienced a huge change with the arrival of thousands of migrants seeking land along road corridors (Almeida 1992). At the same time, leaders from Acre seeking to attract investors gave presentations in southern Brazil, extolling opportunities in Acre via the cheap land available (Bakx 1988).

Such propaganda attracted numerous investors who bought up old rubber estates to be cleared for cattle ranchers. Colonization, ranching and land speculation therefore resulted in extensive deforestation in Acre. The shift in land use became a salient public issue, as local and national newspapers reported on Acre as a new cattle-raising frontier (O Rio Branco 1977, Correio Agro-Pecuário of São Paulo 1981). For example, the journal Elatômeros (1977) highlighted in an article titled "pasture takes over rubber places" that a new agroindustry was emerging in Acre.

However, rubber estates were not simply empty spaces awaiting new owners; rather, they still harbored communities of forest extractivists ("rubber tappers"). These pre-existing populations, though perhaps forgotten by rubber estate owners and overlooked by the newly-arrived ranchers, nonetheless sought to defend their long-standing claims to use the forest. The arrival of ranchers thus created a fundamental conflict over land use between forest extractivism, which required conservation of standing forests, and cattle ranching, which by definition required deforestation.

The ongoing conflicts resulted in periodic high-profile assassinations of labor leaders in eastern Acre, most notably the murder of Chico Mendes in December 1988. That year was also a year of record deforestation and burning, and the correspondence of deforestation and human rights abuse in Acre, drew the attention of international environmental as well as human rights organizations.

This stimulated action by the Government of Brazil, which led to the creation of federal Extractive Reserves, an alternative land use model that called for both the conservation of forests and sustainable use by local people. The Chico Mendes Extractive Reserve was among the first created by federal government, and the number of extractive reserves in Brazil has since risen rapidly (Gomes 2009).

Official recognition of extractive reserves stimulated further innovations in land tenure models in the Brazilian Amazon. By the early 2000s, the federal government as well as the Government of Acre had formalized various land tenures models ranging from private individual properties to extractive reserve, agricultural settlements, agro-extractive settlements, and agroforestry poles , each with distinct rules concerning deforestation. The Government of Acre's Ecological-Economic Zoning Plan (Governo do Acre 2006) calls for distinct land users across the state, depending on natural resources, historical resource management by local populations, and other considerations. The zoning plan's maps are a multi-colored mosaic of different types of land, often side by side, with different use rules and deforestation limits.

In this very changed context, Acre has incurred a new generation of infrastructure initiatives. In the 1990s, the Government of Brazil pursued a series of integration initiatives that called for (among other things) paving of highways in the Brazilian Amazon. Among other such projects, the BR-317 Highway which runs through eastern Acre was paved by the end of 2002. Further, in 2000, the Government of Brazil and several other governments of countries in South America consolidated a series of agreements for trans-boundary infrastructure projects, under the Initiative for the Integration of Regional Infrastructure in South America, or IIRSA (IIRSA 2008).The first phase of IIRSA was the five-year period from 2006-2010, and highlighted the "Brazil-Peru axis" in the southwestern Amazon including Acre. This re-consecrated BR-317 as the Inter-Oceanic Highway by funding paving of that road into Peru and to the Pacific coast, thus facilitating access from Acre to the Pacific rim markets.

Given the new infrastructure alongside diverse tenure arrangements, three key questions arise. The first is whether deforestation accelerated during and/or after highway paving. While opening roads for truck transport certainly facilitates accessibility and marketing of products (Nelson et al. 2004; Pfaff et al. 2007), highway paving provides additional benefits. This is

especially likely to be the case in the Amazon, as paving permits year-around transport, whereas unpaved roads may become impassable in the rainy season.

Second, the impacts of paving are unlikely to be the same everywhere, since the benefits of accessibility are not equally distributed in space around a highway corridor. Specifically, lands more distant from the Inter-Oceanic Highway are likely to exhibit lower deforestation increments during paving than more accessible lands closer to the highway. This is because paving the highway did not also involve paving of secondary roads, which continue to pose limitations on accessibility that grow with distance from the highway itself. Hence there are both spatial and temporal dimensions to deforestation in regions receiving new infrastructure.

And third, many theoretical frameworks for understanding land cover change highlight the importance of institutions, notably land tenure, as a key determinant (Geist and Lambin 2006). Insofar as different lands have use rules with differing limitations on deforestation, there should be distinct deforestation levels and dynamics. This should be the case even for adjacent lands if they have different use rules, and for lands close to the highway where accessibility is high. Hence, to the extent that land users actually follow use rules, we should also expect that deforestation dynamics vary according to land tenure rules, regardless of highway paving and distance from highway. Lands with more restrictive rules concerning deforestation should exhibit less deforestation.

Land Tenure Types in Eastern Acre

The foregoing discussion requires further elaboration on the specifics of use rules in different land tenure types found in eastern Acre. This section therefore provides an overview of the key tenure models I will compare in this chapter, highlighting the origins, goals, use rules, and deforestation limits. Specifically, I will compare 1) Agricultural Settlement Projects (Projetos de Assentamento, or PAs), 2) Directed Setlement Projects (Projetos de Assentamento

Dirigido, or PADs), 3) the Chico Mendes Extractive Reserve (Reserva Extrativista Chico Mendes or RESEX, 4) Agro-extractive Setlements (Projetos de Assentamento Agroextrativista, or PAEs), and 5)Agroforestry Poles (Polos Agrofloretais, or PEs).

Agricultural Settlements Projects (PAs) were created in 1987. Their legal basis is founded in the guidelines of the First National Plan for Agrarian Reform which has as its mandate the design of land settlements that serve the "social function" of rendering land productive via agriculture to provide livelihoods by feeding populations. The creation of a PA emphasizes steps to identify productive land users who will benefit from agriculture. PAs generally involve small landholdings of 20-100 ha titled to families that engage in agricultural land use. PAs carry deforestation limits of 20% for landholdings of 100 ha or more.

Directed Settlement Projects (PADs) are settlement projects designated to implement a social system that allows a sustainable production, to meet the social function of land and economic production as well as the social and cultural rights of the rural families (ZEE 2006). This type of project was created by INCRA (National Institute of Colonization and Agrarian Reform) in the late 1970s with the objective to fulfill the regularization of land parcels over the domain of the federal government. It was also designed to provide opportunities for small farmers to grow food crops and perennial trees and animals. However, cattle ranching is extensively practiced, causing serious land degradation and deforestation. PADs have family lots of similar size to PAs and the same deforestation limits as PAs, i.e., 20%.

The Chico Mendes Extractive Reserve (CMER) was created in 1990 in response to the land conflicts discussed above. The CMER is divided into roughly 50 rubber estates (seringais), each of which has approximately 20-50 family settlements (colocações). Each family settlement has 4-6 rubber trails, covering roughly 100 ha, so each family has use rights to roughly 400-600 ha. Like other RESEX, CMER lands are federal lands, and resident families gain use rights via

concessions based on traditional occupation in the past. Use rights are defined based on the CMER management plan, jointly defined by residents and IBAMA (Brazil's Institute for the Environment, now the Chico Mendes Institute, IMC), part of the federal government. The CMER and other RESEX are thus managed as conservation areas, and have restrictions on resource use including 10% limit on deforestation and a 5% limit on pasture area.

Agroextractive Settlement Project (PAEs) are a special type of settlement where the activities to be developed are based on extraction of forest resources. In the Amazon region, due to great concern about preserving the forest, PAEs developed in order to take into account the characteristics of the traditional rubber tapper in the region. PAEs are similar to RESEX, as both have 10% deforestation limits. Both are federal lands where residents have secure land use rights. Both are managed by the settlers who receive concessions with use rights. In order to obtain concessions, settlers need to follow the use rules established by management plans. However, PAEs are administered by INCRA and are not designated as conservation units.

Agroforestry Project **(**PEs) constitutes an alternative modality of settlement located near cities. PEs were designed for occupation by jobless former small farmers and rubber tappers. Lots in PEs are typically small (3-10 ha) and intended for intensive use as via gardening and agroforestry. PEs are also known as green belts and are managed by the state government of Acre, though PE regulations are guided by INCRA requirements. Because PEs have small lots, they do not have deforestation limits.

Methods: Study Area

The state of Acre covers an area of 164,220 km^2. It is located in the western Amazon (7° 07´ - 11° 08´ S and 66^o30´- 74° 00´W), along the southeast and northwest boundaries of the state of Amazonas, west and south of Peru, south and southeast of Bolivia, and southeast of Rondônia state. The study site is located in the southeastern part of Acre, Brazil and encompasses the upper

and lower Acre river basin (Figure 3-1). I focus on eastern Acre because it encompasses the Inter-Oceanic Highway route thru Acre and it contains numerous different land tenures types, including several examples of each of the types I will compare in this chapter. (Figure 3-2).

Specifically, eastern Acre encompasses 14 PAs, 2 PADs, 5 PAEs, 5 PEs and 1 RESEX all of which are included in the analysis in this chapter. Tables 3-1, 3-2, 3-3, 3-4 and 3-5 list the PAs, PADs, CMER, PAEs, PEs by name, municipality, date of creation, land area, number of families settled and families per km_2. The settlement areas in the analysis not only encompass a range of tenure rules, but also varying distance from the Inter-Oceanic Highway.

The analysis uses Landsat imagery for 1986, 1991, 1996, 2000 and 2005. Landsat ETM and TM images were acquired from the University of Florida. I focus on the dry-season imagery, specifically July, with relatively cloud-free images. The Landsat data were radiometrically calibrated, geometrically registered (image to image rectification), normalized for precipitation differences (when necessary), and mosaicked. The details of these processing methods are outlined in Chapter 2.

Following processing, I pursued classification techniques in order to transform the spectral data into earth surface information through the extraction of thematic features (deforestation, land-use and land-cover change). It was removed clouds, shadows and water from each image before classification. Afterwards, a Principal Component Analysis (PCA), image differencing and thresholding method were applied (Varlyguin et al. 2001).

To help with image classification, secondary products were produced like a tasseled cap index (Kauth et al. 1976), a mid-infrared index (Boyd and Petitcolin 2004), and a 3-by-3 moving window calculation of the variance of each pixel for bands 4, 5 and 7 (the near and mid-infrared bands). Because the visible and thermal bands contained striping, a rule-based or decision tree classification was applied (Breiman 1984) instead of traditional unsupervised or supervised

techniques. The rule-based classification provided flexibility to eliminate visible and thermal bands, using only the near-and mid-infrared bands along with secondary derived products. Field work to support visual interpretation was conducted to help classify land cover and improve classification accuracy. Therefore field visits were conducted to collect training samples for accuracy assessment of forest-non-forest (F-NF) classifications. Land cover was classified into forest, pasture and bare-built, and then pasture and bare-built were aggregated to create the non-forest class.

A forest and non-forest trajectory image was created from the classifications. Image change trajectories are defined as sequences of successive changes in land cover types providing information on changes between two or more time periods of an area or region. Image trajectories for this paper were defined according to Petit et al. (2001) as:

$$m_t = m_c^t$$

Where *mt* is the number of change trajectories, *mc* is the number of land cover classes defined, either forest or non-forest, and the superscript *t* is the number images. Because there are two land cover classes and five observation dates, this resulted in 32 possible trajectories.

For the purpose of the analysis, I divided the study area into 5 segments based on the timing of highway paving, distance from highway, and land tenure type. Figure 3-3 provides a visual overview of these various divisions.

I define paving status based on the year of completion of paving. I divided eastern Acre into five distinct highway segments which were paved at different times. The 5 segments according to paving status are: paving by 1984 (Rio Branco - Senador Ghiomard), paving by 1996 (Senador Guiomard - Capixaba), paving by 1997 (Capixaba – Xapuri), paving by 1999 (Xapuri – Epitaciolandia) and paving by 2002 (Brasiléia- Assis Brasil).[1] I defined land as falling

along a highway segment based on perpendiculars from the road corridor, with interpolations at points among segments with turns in the road. Highway segments thus differ in terms of the timing of paving, permitting a temporal analysis of deforestation before and after paving. The temporal analysis thus compares deforestation before and after paving in each road segment. Figure 3-4 shows the road segments and deforestation by time periods. I anticipate that there will be less deforestation before than after paving, and that deforestation will accelerate after paving. I also anticipate that the effects of paving on deforestation will be greater in areas receiving paving earlier, and in later time periods when paving of the entire corridor neared completion.

In addition, to permit evaluation of the effects of distance from the highway, I created distance buffers along the BR-317 corridor. I defined three distance intervals: 0-<5 km, 5-<10 km, and 10-<15 km. Figure 3-4 shows the buffers along the highway in eastern Acre. While one might define other buffers, these distinctions permit comparisons of lands closer to and farther from the highway. This permits a spatial analysis of deforestation. I anticipate that deforestation will be greater in lands closer to the highway. When combined with the temporal analysis afforded by the definition of highway segments with different paving dates, we can also observe the interaction of the timing of paving with distance from the highway. I expect the effects of paving to be greater in lands closer to the highway.

Finally, I will conduct the land tenure analysis by comparing lands with different use rules and deforestation limits. To that end, I will compare the different land tenure types discussed above, including PAs, PADs, the CMER, PAEs, and PEs. I anticipate the highest deforestation percentages in PEs (which have no deforestation limit), moderately high deforestation in PAs and PADs (which emphasize agriculture and have 20% deforestation limits), and low deforestation in PAEs and the CMER (which emphasize forest extractivism and have 10% deforestation limits). To the extent that these tenure units are located along the

highway, the analysis will also evaluate the interaction of timing of paving and tenure rules. I expect that lands with lower deforestation limits will have lower deforestation, but nonetheless within category of tenure type, deforestation will be greater in those lands where there was earlier paving. And insofar as different land tenure types occur at varying distances from the highway, I will evaluate the interaction of tenure and distance. I expect that lands with lower deforestation limits will have lower deforestation, but within those lands, areas closer to the highway will have greater deforestation.

Given these three explanatory factors for deforestation, the analysis focuses on specific areas along the Inter-Oceanic Highway in Acre. To compare deforestation by time of paving, I focus on the various segments with different years when paving was completed; to evaluate the effects of distance from the highway, I focus specifically on the land areas within the buffers; and to assess tenure differences, I target lands with the tenure designations noted above. In particular, I note that the analysis only focuses on those lands within the tenure units considered that also fall within the distance buffers discussed above. Hence the analysis does not necessarily consider all of the land within a given tenure unit, only those lands within that unit also within the distance buffers. This is necessary to permit direct comparisons of deforestation among lands by each of the three explanatory factors.

Results and Discussion

The analysis proceeds in several steps, each of which involves comparisons among categories of the explanatory factors. All steps in the analysis also compare changes in deforestation over time from 1986 to 2005. First, I evaluate deforestation by time of highway paving by comparing the road segments from Rio Branco to Assis Brasil. Second, I analyze deforestation by distance from the Inter-Oceanic Highway using the distance buffers. Third, I simultaneously consider both time of paving and distance from the highway. This permits

multivariate analysis to see if one explanatory factor affects the impact of another on deforestation. Specifically, the analysis considers whether deforestation increments have been faster in buffers closer to the highway, but especially so in those lands where the highway was paved earlier. Fourth, I consider the effects of land tenure by comparing the tenure types found in eastern Acre. I then (fifth) review land tenure and highway paving to see if earlier paving yields faster deforestation equally for each type of tenure unit. And finally (sixth) I conclude the analysis by presenting findings on tenure and distance from the highway, in order to assess whether distance also modifies the effects of tenure such that deforestation is greater closer to the highway for all tenure types.1

Time of Paving and Deforestation

Figure 3-5 presents findings for deforestation in the selected lands in eastern Acre from 1986-2005, broken down by time of highway paving. Overall, deforestation rose from roughly 4% in 1986 to approximately 28% in 2005. The trajectory of deforestation shows non-linearities, with an increase from 1986-1991, a deceleration from 1991-1996, and an acceleration from 1996-2000 and especially during 2000-2005. The accelerations during the last two periods coincide with a major effort to complete paving of the Inter-Oceanic Highway in Acre during the late 1990s and early 2000s.

Figure 3-5 also presents deforestation estimates for each highway segment. In general, areas with earlier paving have more deforestation. The Rio Branco-Quinari (Senador Guiomard) segment, which was paved by 1984, exhibits the highest deforestation percentages. Moving south and west along the Inter-Oceanic Highway, road segments were paved more recently, and each progressive segment exhibits lower deforestation percentages. Rio Branco-Quinari has a

[1] The analysis does not take the next step, which would simultaneously consider the effects of all three variables. The reason for this is because such a table would be huge and would have too many cells without data, which would undermine comparisons.

higher deforestation percentage than Quinari-Capixaba, which exhibits more proportional deforestation than Capixaba-Xapuri, and so on. The one exception to this pattern is the last segment, Brasiléia-Assis Brasil, which has a higher deforestation percentage than many of the other segments. Hence the gradient holds in most but not in all cases.

Another way to evaluate highway paving and deforestation in Figure 3-5 is to compare deforestation percentages before and after paving by road segment. For Rio Branco-Quinari, the highway was paved before the first observation in 1986. I note that deforestation in 1986 was low (under 5%) but it rose very rapidly thereafter, exceeding 60% by 2005. Hence in this segment, there was little deforestation by 1986 but rapid forest loss after road paving. The question then is whether other road segments also exhibit little deforestation before paving and rapid forest loss afterwards as well. Here the findings are somewhat mixed. The Quinari-Capixaba segment was paved by 1996, and exhibits little deforestation up to 1996 (roughly 11%), but there is little response up to 2000; however, deforestation there jumped to almost 35% by 2005. The next segment, Capixaba-Xapuri, exhibits a similar pattern of a delayed jump in deforestation after paving. Xapuri-Epitaciolândia also had little deforestation by 2000, by the time paving was completed, and deforestation rises faster thereafter. Brasiléia-Assis Brasil received paving by 2002, but deforestation was already rising in the late 1990s. These findings do suggest a correspondence in the timing of paving and accelerated deforestation, but they also suggest that other processes are also at work, since some segments have delays in the acceleration of deforestation while others exhibit rising deforestation during paving rather than only afterwards.

Distance to Highway and Deforestation

If the timing of paving only partly accounts for deforestation patterns, it is also possible that the effects of the Inter-Oceanic Highway reflect proximity of land to the highway itself. I

therefore compare lands at different distances from the highway, in buffers with 5 km intervals. Figure 3-6 presents deforestation along the Inter-Oceanic Highway in eastern Acre from 1986-2005 by the distance buffers.

There is considerable prior literature on land use and land cover change, including for the Amazon, which emphasizes the importance of accessibility for forest loss, and consistently shows greater forest loss closer to highways. Figure 3-6 is therefore surprising, because it shows rather similar deforestation percentages among the buffers out to 15 km from the Inter-Oceanic Highway. Initially, deforestation shows a U-shaped distance curve, but with time, the standard distance gradient emerges, such that deforestation is greater closer to the highway by 2005. Nonetheless, there is not a large difference between the 5-10 km and 10-15 km buffers. For the most part, access to land is not easy far from the highway; secondary roads in rural areas of Acre remain unpaved and virtually impassable during the rainy season. One reason the deforestation percentages at 10-15 km appear high (Figure 3-7A) is that the capital city of Rio Branco falls within that buffer as the Inter-Oceanic Highway passes around the center of town.

One way to evaluate that possibility and control for the effects of Rio Branco is to break down deforestation by both timing of paving (and thus road segment) as well as distance from the highway. Figure 3-7A-E provides this multivariate breakdown.

Figure 3-7A confirms that the Rio Branco-Quinari segment exhibits higher deforestation percentages in the buffers more distant from the Inter-Oceanic Highway. This is true of all time points, and is consistent with the interpretation that the higher deforestation percentages in the most distant buffer are due to urban growth of Rio Branco. By contrast, the other road segments Figure 3-7B-E all exhibit deforestation distance gradients that correspond to previous work on accessibility and land cover change. Further, the distance gradients become stronger as one moves farther from Rio Branco. Whereas the distance gradient is weak in the Quinari-Capixaba

segment Figure 3-7B, it is stronger in all three of the other highway segments Figures 3-7A-E, and in each of those segments, the distance gradient appears at all time points. Hence distance gradients in deforestation do have a relationship with time since paving, but the relationship is distorted in the study region due to the routing of the Inter-Oceanic Highway around Rio Branco.

Land Tenure Type and Deforestation

Figure 3-8 presents deforestation estimates for lands in the tenure categories found along the Inter-Oceanic Highway in Acre. Based on the foregoing discussion of these tenure types, I expect deforestation to be highest in the agroforestry poles (PEs) which lack deforestation limits, followed by the PAs and PADs (which have 20% deforestation limits), and then the PAEs and the CMER (which have 10% deforestation limits). Figure 3-8 shows large differences in deforestation percentages among the tenure categories, as well as distinct deforestation trajectories over time. In 1986, the PEs indeed had the highest deforestation percentage, followed by the PAs and PADs, and then the PAEs and CMER. However, over time, deforestation rose faster in the PAs and PADs than in the PEs, and by 2005, the PAs and PADs had the highest deforestation percentages.

The higher rates of deforestation in the PAs and PADs are explained by the expansion of cattle pasture. Crop prices in Acre have fluctuated over time, whereas cattle prices have proven relatively stable or rising. At the same time, PEs exhibited deforestation fluctuations, notably a decline after 1991 before a rise after 2000, in part due to instabilities in crop marketing opportunities. Families in PAs and PADs faced the same difficulties, but had more land that permitted pasture expansion. That said, Figure 3-8 also indicates that at least in lands close to the Inter-Oceanic Highways in PAs and PADs, deforestation exceeds the legal limits. This may be due to properties under 100 ha, which are exempt from the 20% rule, or “grandfathering” in older PADs which existed before the 20% rule.

In any event, there are lower estimates among PAE and the CMER. This does not change over time. That is not to say that deforestation does not increase over time in these tenure categories; both exhibit deforestation of roughly 1% in 1986 and 9% in 2005. What is remarkable is how similar the trajectories of the two categories are; both have 10% limits and both have virtually identical trajectories. Further, both types remained under their legal deforestation limits as of 2005.

While land tenure clearly matters for deforestation trajectories, it was also clear in earlier tables that time since paving and distance to highway are also important. Figure 3.9A-E therefore compares deforestation trajectories among both land tenure types and highway segments that were paved at different times. Overall, deforestation estimates decline as one move from highway segments with earlier paving to later paving, regardless of land tenure type. This applies to the PEs as well as the PAs and PADs, and also the PAEs and the CMER. While tenure type does affect the level and rate of deforestation, it does not modify the effect of highway segment, in that lands along highway segments with more recent paving have less deforestation, regardless of tenure type.

There are however some non-linearities. For example, among the PAs, deforestation is relatively high in the Rio Branco-Quinari segment, low in the Quinari-Capixaba segment, and moderate in the segments beyond. One explanation for this is that in some cases, there are older settlements in areas with more recent paving, and settlement age may be countering the effects of paving recency. Among the PAs, near Capixaba, there has been an emergent agribusiness enterprise for sugar cane. Investment for sugar cane processing began in 1989 with the creation of Brazilian Alcohol S/A, (ALCOBRAS), which was linked to Brazil's National Alcohol Program, PROALCOOL. ALCOBRAS however went bankrupt. Since then, the land and what remained of the machinery were abandoned, resulting in expropriation of the area. The

ALCOBRAS area was divided in two new PAs, PA Alcobras and PA Zaqueu Machado. The bankruptcy and relatively recent creation of these PAs help explain the relatively low deforestation in this highway segment.

Another issue in Figures 3- 9 is that not every tenure type occurs in every highway segment. The PAs and PADs Figure 3-9A-B tend to occur in older segments closer to Rio Branco, and the PEs, PAEs and the CMER Figure 3-9C-E are located in newer segments farther away. This might lead one to suspect that tenure differences are actually due to these different spatial distributions of the tenure types, as where PAs and PADs have more deforestation merely because they are in areas with older paving than PAEs and the CMER. This is however not the case: these two groups of tenure types do occur in some of the same road segments and in those segments, such as Capixaba-Xapuri and Xapuri-Epitaciolandia, PAs and PADs still have greater deforestation than PAEs and the CMER. Further, the spatial coincidence of PAs and PADs permits more direct comparisons of their deforestation trajectories; the same is true for PAEs and the CMER.

One final topic that merits comment is that the acceleration in deforestation from 2000 to 2005 occurs across all tenure categories and road segments, regardless of the combination of these characteristics. This suggests a generalized shift in land cover during this period, possibly due to the conclusion of road paving, and/or acceleration in economic growth. In any case, local circumstances tied to time since paving or tenure type do not greatly modify the fact that deforestation percentages roughly double in all lands in this analysis from 2000 to 2005.

Figure 3-10 presents a similar multivariate analysis by comparing deforestation percentages over time among land tenure types and distance from the Inter-Oceanic Highway. We know that because the highway passes around Rio Branco, the distance gradient in deforestation does not appear there; and we know that PAs and PADs tend to occur in highway

segments closer to Rio Branco. Hence it is not surprising that deforestation either does not decline much by distance (PAs) or it actually rises (PADs). Conversely, distance gradients in deforestation do appear for the other tenure types that tend to occur along other highway segments, as for PEs, PAEs, and the CMER. Hence observation of gradients in deforestation by distance from the Inter-Oceanic Highway is affected by the route of the road and thus road segment, more than land tenure per se. If we compare the lands in different tenure types unaffected by Rio Branco, i.e. the PEs vs. the PAEs and CMER, we find that they all exhibit distance gradients with lower deforestation percentages in buffers farther from the highway. This despite the fact that the PEs have very different land use rules than the PAEs and CMER.

Conclusion

Several key conclusions arise from the foregoing analysis. First, deforestation along the Inter-Oceanic Highway has risen over time, and accelerated around the time that paving concluded in Acre. There is some evidence of non-linearities, with a slowdown in the early 1990s, followed by an exponential acceleration in the late 1990s and early 2000s. This finding applies regardless of time of paving, distance from the highway, or land tenure type. The acceleration is likely to be related to the conclusion of paving of the highway corridor, but paving by itself is unlikely to be a sufficient explanation; more generalized economic expansion is also key, with the paved highway facilitating new land use and marketing.

That said, there are spatial differences as well as temporal dynamics at play in land cover change in eastern Acre. All three of the explanatory factors I considered – time since paving, distance from the highway, and land tenure type – exhibit important effects on deforestation percentages. Lands along highway segments with earlier paving, closer to the highway, and with tenure rules including higher deforestation limits (or no limits at all) all exhibited higher deforestation percentages. Further, with one exception, there were not strong interactions among

these explanatory variables. For example, lands along highway segments with earlier paving had higher deforestation regardless of distance from the highway; and lands along segments with older paving also had greater deforestation regardless of land tenure type. The interaction concerned highway segment and distance from the highway, and arose due to the route of the Inter-Oceanic Highway, which skirts around the city of Rio Branco. As a result, the distance gradient disappears or even reverses in highway segments with earlier paving, but in segments farther from Rio Branco, the gradient appears normal, with less deforestation farther from the highway. Overall, the analysis confirms the importance of time since paving, distance from highway and land tenure type, and suggests that interactions among these factors are not strong, with an exception based on the route of the highway.

Hence there are important temporal non-linearities in deforestation in eastern Acre, as well as substantial spatial differences in deforestation. Future research can further inquire into these temporal dynamics and spatial patterns. For one thing, it will be important to know if the acceleration in deforestation since 1996 continues beyond 2005. If that is the case, it is likely that several land tenure types will exhibit deforestation beyond their legal limits. PAs and PADs, or at least the portions thereof included in this analysis, already exceeded their deforestation limits in 2005; PAEs and the CMER were nearing their limits by then as well.

Another issue would be to expand the distance buffers to consider lands farther out from the Inter-Oceanic Highway. This analysis went out to 15 km, but previous research suggests that road impacts extend to 50 km. Hence future research could increase the buffers out to 50 km, or even beyond. This would expand the land areas under analysis, but come at the cost of having many more distance buffers to compare. It would also be useful to identify the urban area of Rio Branco and mask that out from the analysis, since the deforestation analysis is primarily concerned with rural land use. However, this is only likely to partially correct for the impact of

the state capital; even beyond the urban boundaries but close to town, land use is likely to be intensive and forests are likely to be scarce.

A key policy implication of this analysis concerns land tenure and regional integration via highway paving. Clearly, land tenure matters for deforestation levels, suggesting that landholders at least try to follow established land use rules. That said, there is some evidence of rule-breaking, particularly at a time when highway paving was concluding in eastern Acre, when deforestation accelerated. Under those circumstances, differences in deforestation due to tenure remained, but levels of deforestation with completed highway paving were nearing or exceeding legal limits. Hence highway paving and the economic growth it is intended to facilitate is likely to come into conflict with land use rules. This raises issues of land use alternatives in a context where cattle ranching is a pre-eminent land use and requires extensive pasture areas (Hoelle 2011; Gomes 2009). Despite many innovative policy programs seeking "neo-extractivism" (Rego 1999) and other forms of diversified and intensified land use (e.g., Maciel 2006), there remain market pressure and a favorable political economy for cattle in Acre (Walker, et al. 2009). Whether new policy proposals or market shifts can obviate the conflicting pressures on land use between tenure rules and market access remains to be seen, but is now a crucial issue along the Inter-Oceanic Highway corridor in eastern Acre.

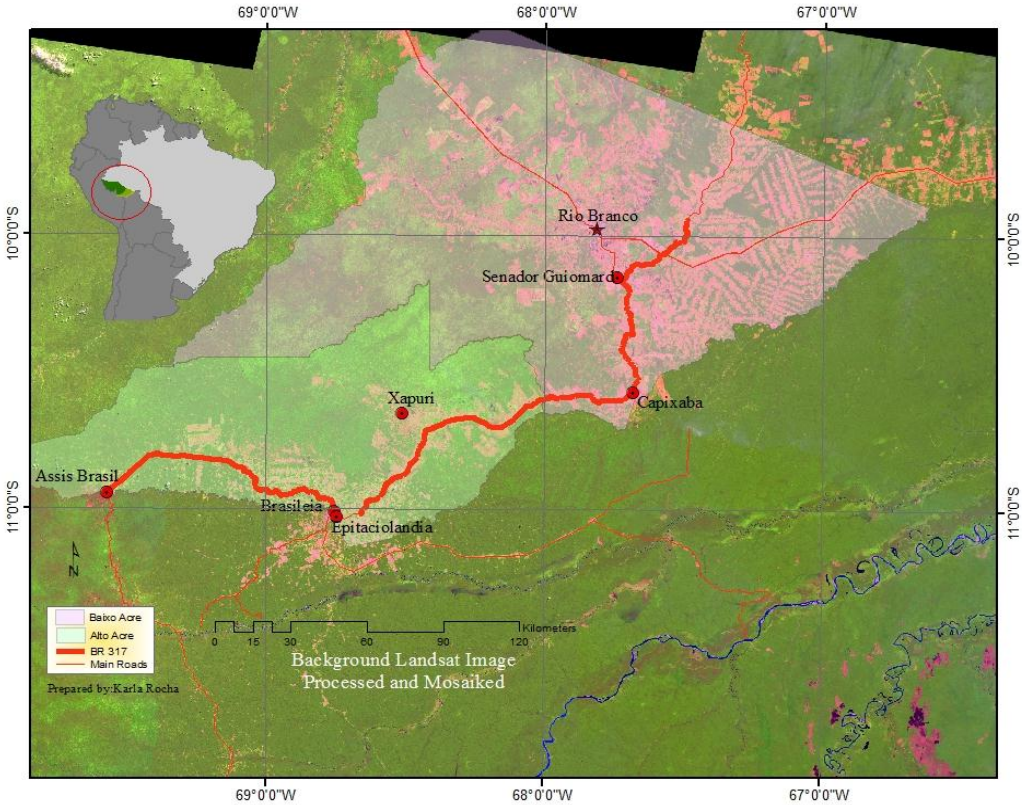

Figure 3-1. Study area showing the lower and upper Acre river basin.

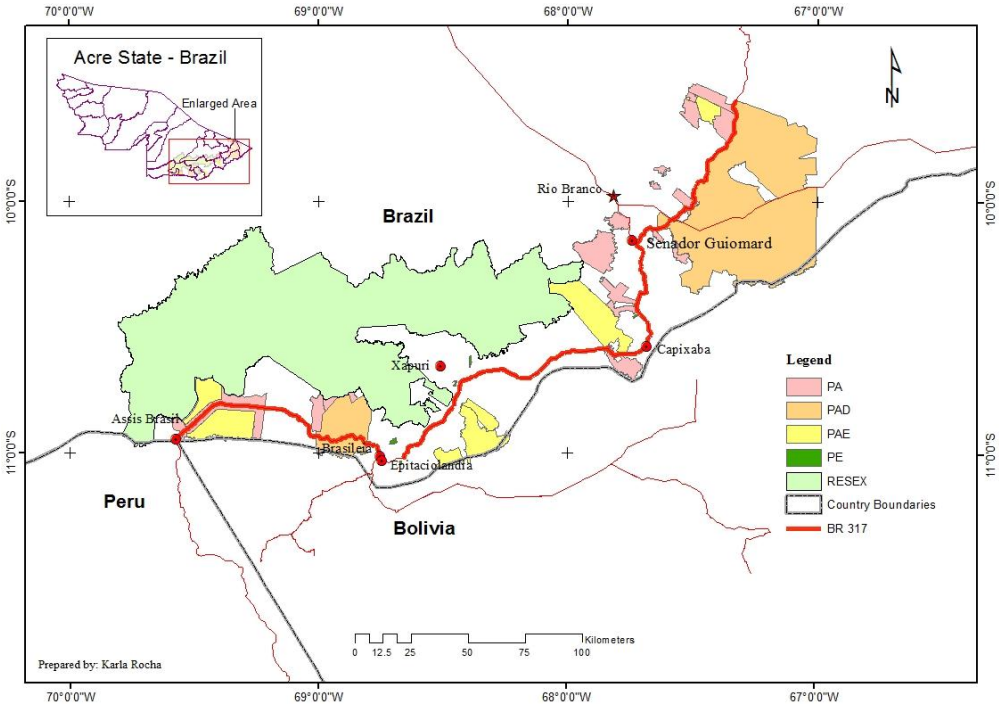

Figure 3-2. State of Acre and its different land tenures category.

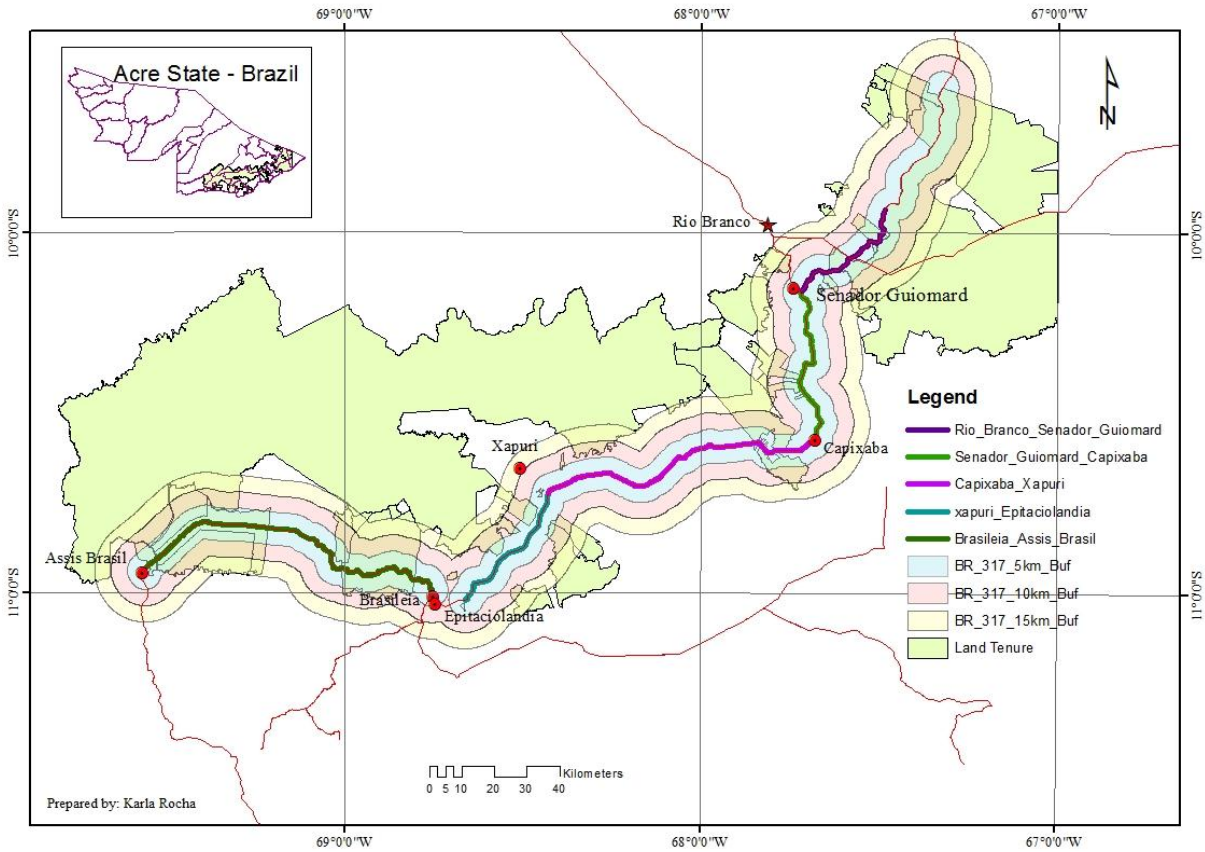

Figure 3-3. Showing road segment according to paving status, 5, 10 and 15 km buffer and land tenure category.

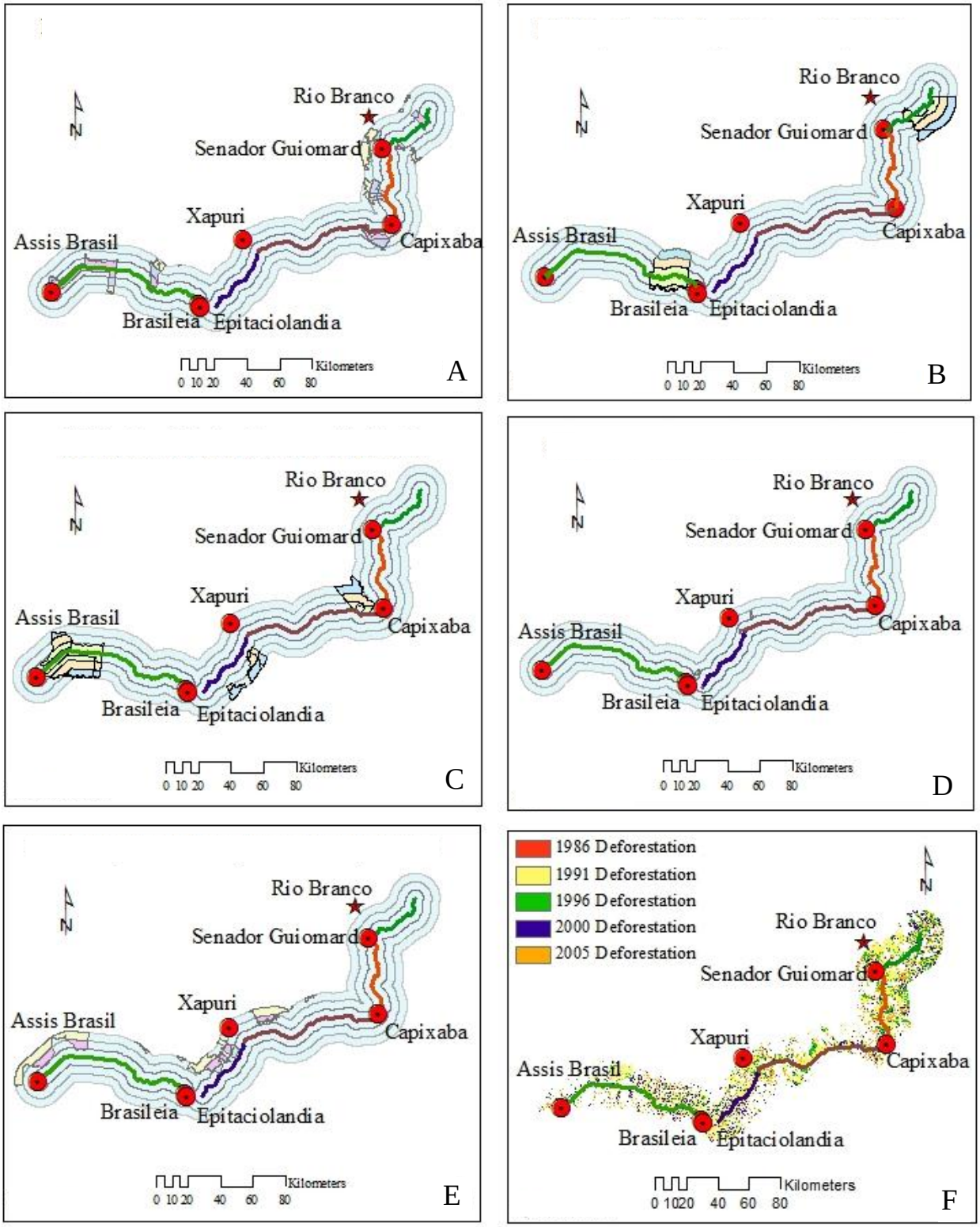

Figure 3-4. Tenure distribution by buffer and paving status. A) Agricultural Settlement Project (PA). B) Directed Settlement Project (PAD). C) Agro-Extractive Settlement Project (PAE). D) Agroforestry Pole (PE). E) Extractive Reserve (RESEX). F) Deforestation through 1986 to 2005.

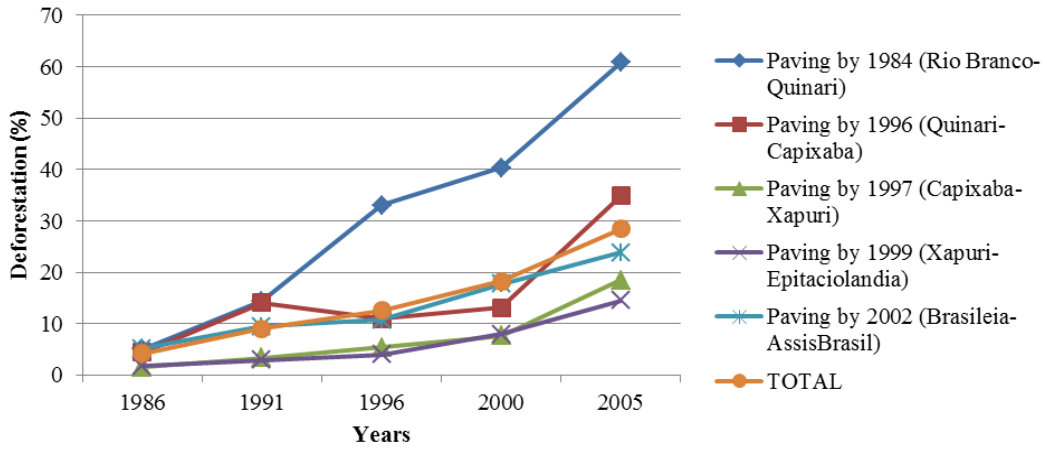

Figure 3-5. Deforestation thru time by highway paving status, selected lands along the Inter-Oceanic highway in Acre, Brazil 1986-2005.

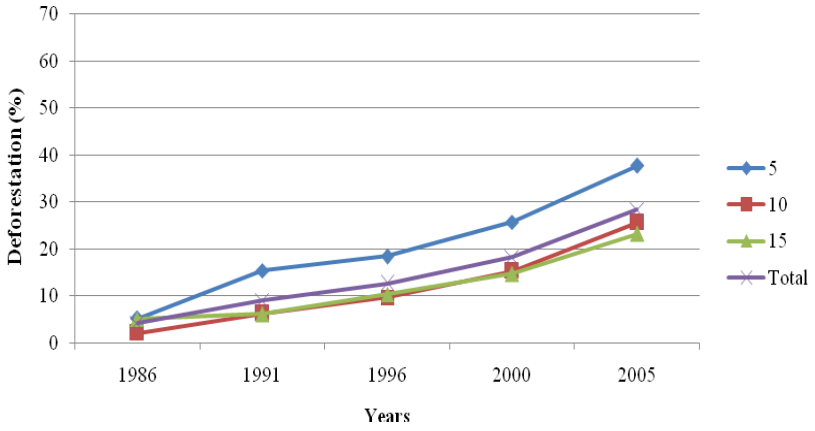

Figure 3-6. Deforestation thru time by distance from highway, selected lands along the Inter-Oceanic highway in Acre, Brazil 1986-2005.

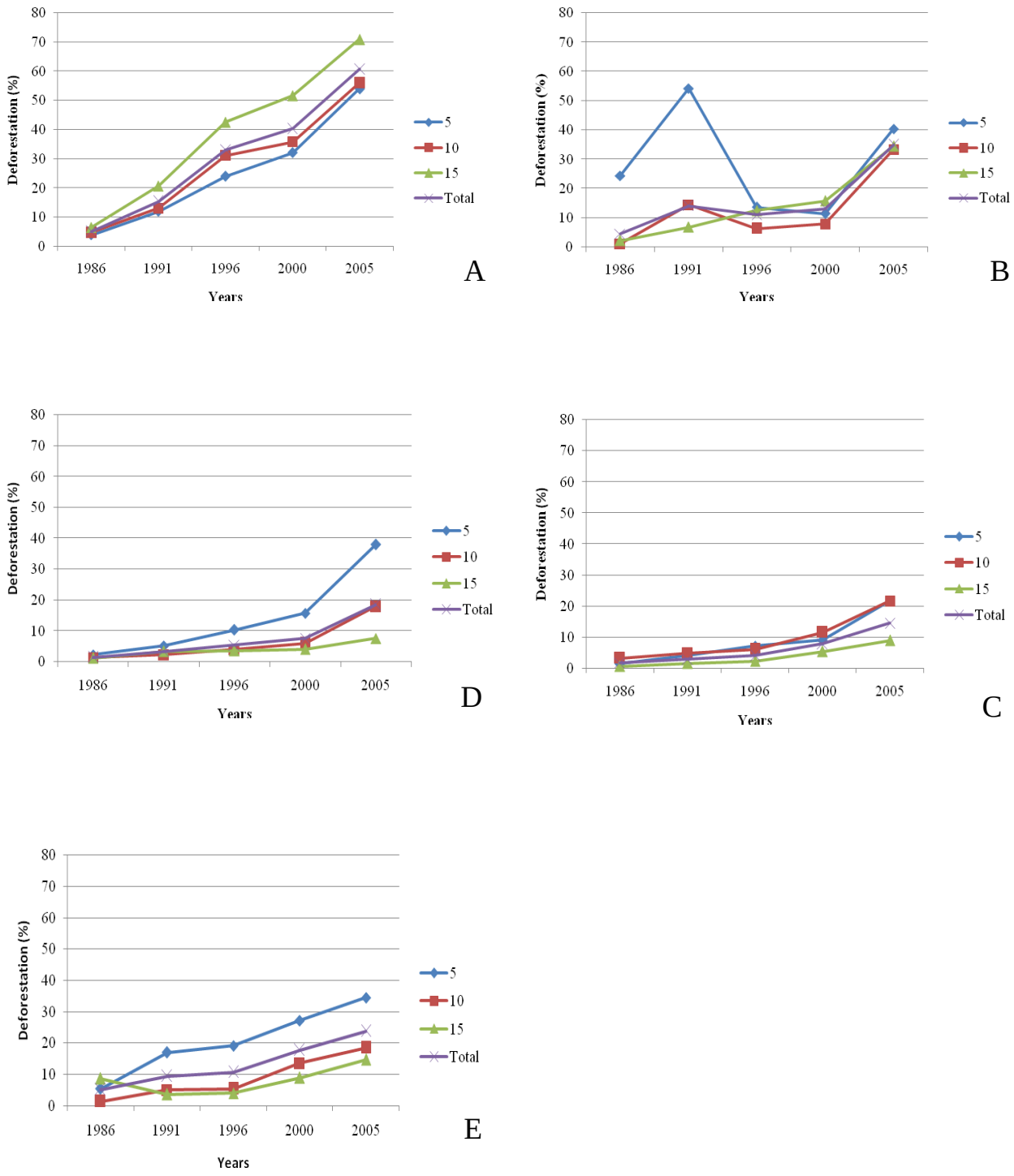

Figure 3-7. Multivariate road breakdown. A) Paving by 1984 (Rio Branco-Quinari). B) Paving by 1996 (Quinari-Capixaba). C) Paving by 1997 (Capixaba-Xapuri). D) Paving by 1999 (Xapuri-Epitaciolândia). E) Paving by 2002 (Brasiléia-Assis Brasil).

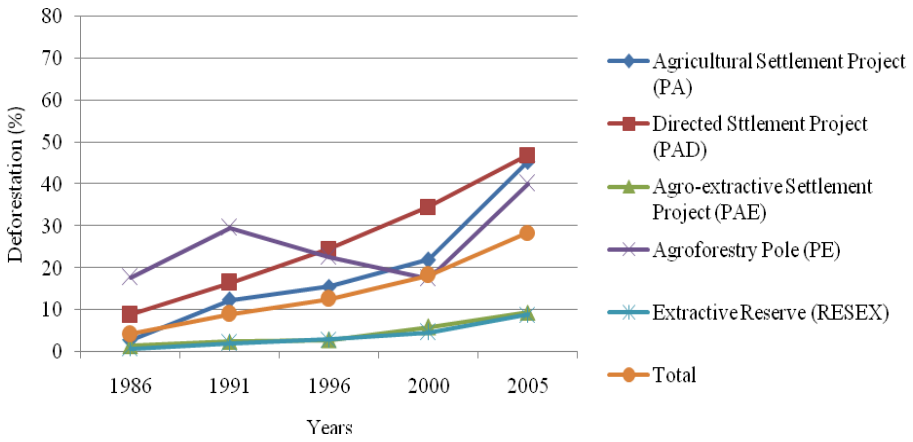

Figure 3-8. Deforestation thru time by land type, selected lands along the Inter-Oceanic highway in Acre, Brazil 1986-2005.

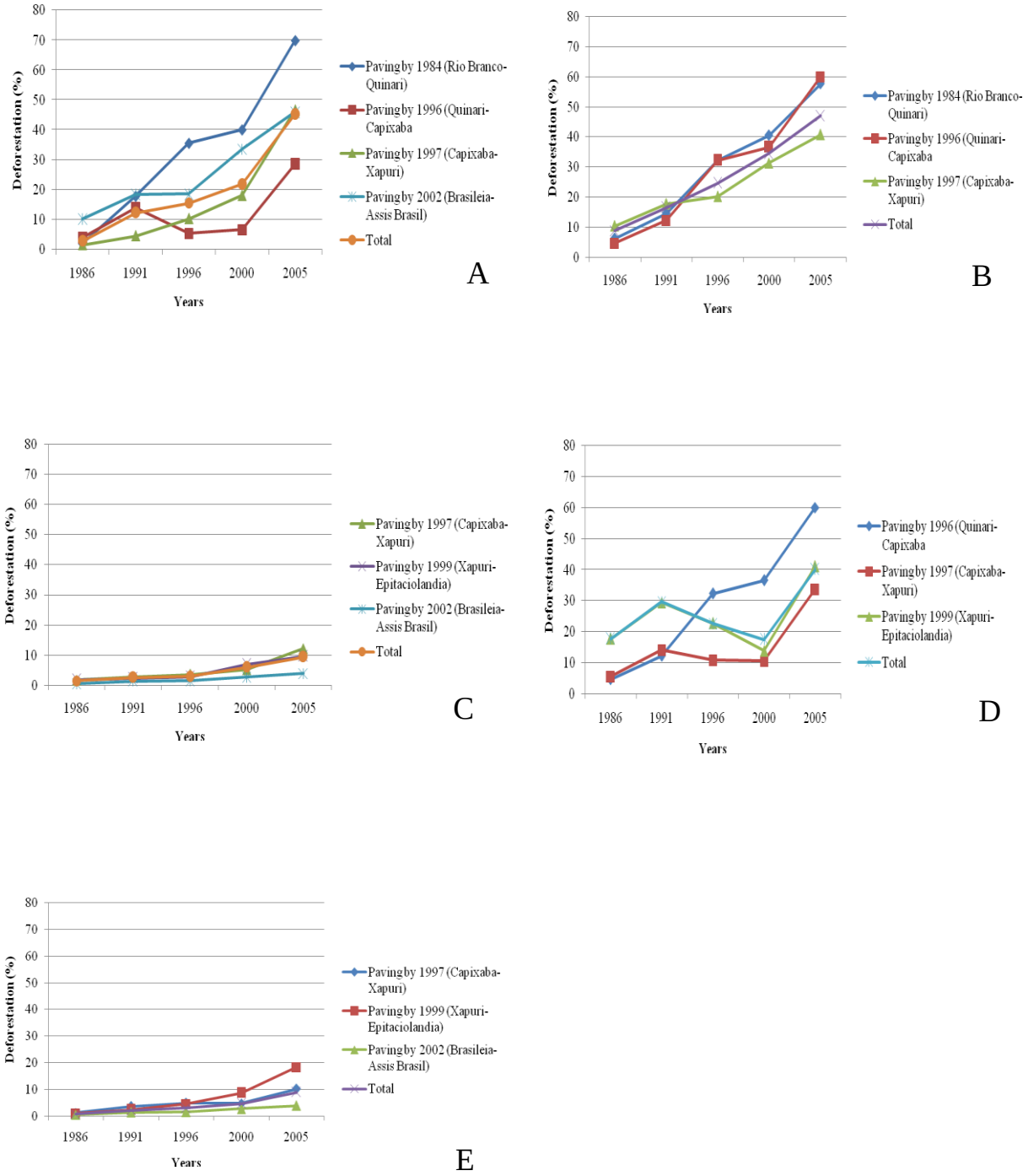

Figure 3-9. Deforestation trajectories by land tenure types and highway segments paved at different times. A) Agricultural Settlement Project (PA). B) Directed Settlement Project (PAD). C) Agro-Extractive Settlement Project (PAE). D) Agroforestry Pole (PE). E) Extractive Reserve (RESEX).

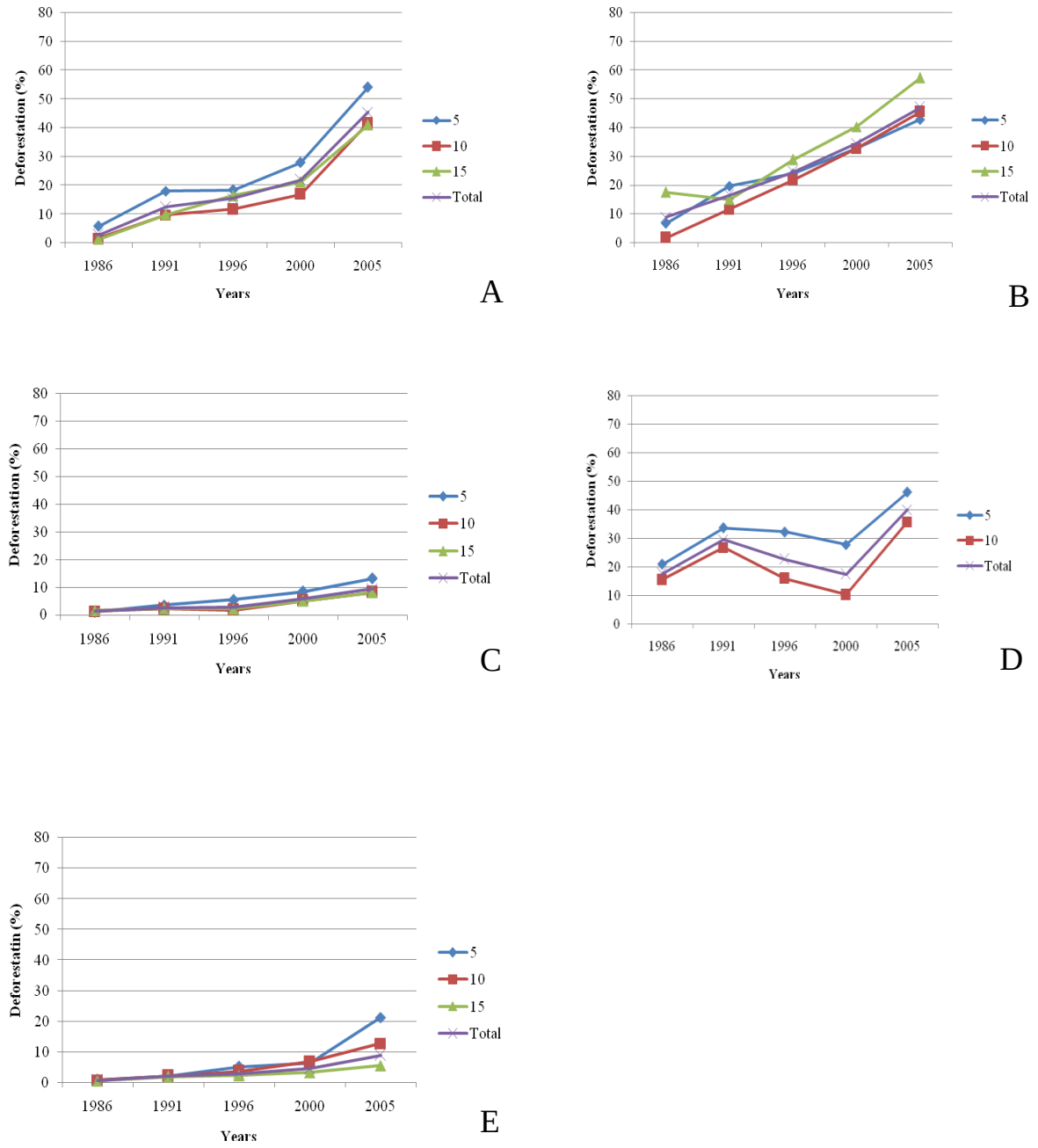

Figure 3-10. Deforestation percentages over time by land tenure types and distance from the Highway. A) Agricultural Settlement Project (PA). B) Directed Settlement Project PAD). C) Agro-Extractive Settlement Project (PAE). D) Agroforestry Pole (PE). E) Extractive Reserve (RESEX).

Table 3-1. Settlement Projects (PAs).

Project	Municipality	Creation date	Area (ha)	Capacity	Families settled	Families/ km_2
Alcobras	Capixaba	1998	7748.68	443	408	5.27
Baixa Verde	Rio Branco	1996	4867.53	165	165	3.39
Benfica	Rio Branco	1994	5391.64	300	300	5.56
Colibri	Rio Branco	1995	1498.06	42	38	2.53
Vista Alegre	Rio Branco	1987	1022.45	35	28	2.73
Moreno Maia	Rio Branco	1997	21337.03	500	475	2.22
Limeira	Senador Guiomard	1998	7551.71	123	130	1.72
Paraguassu	Assis Brazil	2004	3688.69	98	95	2.57
Petrolina	Senador Guiomard	2005	3070.47	85	84	2.73
Pao de Acucar	Brasiléia	1999	7397.35	123	118	1.59
Sao Gabriel	Capixaba	1996	10176.74	161	161	1.58
TresMeninas	Brasiléia	1999	2004.35	61	58	2.89
Zaqueu Machado	Capixaba	2001	3757.14	236	227	6.04

Source: Governo do Acre (2006).

Table 3-2. Directed Settlement Projects (PADs).

Project	Municipality	Creation date	Area (ha)	Capacity	Families settled	Families/ km_2
Pedro Peixoto	Senador Guiomard	1977	357631.30	4727	4654	1.30
Quixadá	Brasiléia	1981	50523.38	1032	1017	2.01

Source: Governo do Acre (2006).

Table 3-3.Chico Mendes Extractive Reserve (CMER).

Project	Municipality	Creation date	Area (ha)	Capacity	Families settled	Families/ km_2
CMER	Xapuri	2003	970570.00	2050	1969	0.20

Source: Governo do Acre (2006).

Table 3-4. Agro-Extractive Settlement Projects (PAEs).

Project	Municipality	Creation date	Area (ha)	Capacity	Families settled	Families/ km_2
Chico Mendes	Epitaciolandia	1989	24932.00	88	88	0.35
Equador	Epitaciolandia	2001	7845.68	36	36	0.45
Porto Rico	Epitaciolandia	1991	7862.05	46	46	0.58
Remanso	Capixaba	1987	43316.34	189	184	0.42
Santa Quiteria	Brasiléia	1988	69015.43	300	289	0.41

Source: Governo do Acre (2006).

Table 3-5. State Agroforestry Projects (PEs).

Project	Municipality	Creation date	Area (ha)	Capacity	Families settled	Families/ km_2
Polo Agrof. Brasiléia	Brasiléia	2001	538.33	74	68	12.63
Polo Agrof. Capixaba	Capixaba	2008	254.46	30	20	7.87
Polo Agrof. Epitaciolandia	Epitaciolandia	2001	129.89	9	8	6.20
Polo Agrof. Xapuri I	Xapuri	2002	364.79	31	30	8.24
Polo Agrof. Xapuri II	Xapuri	2002	231.15	35	35	15.15

Source: Governo do Acre (2006).

CHAPTER 4
MEASUREMENT AND CHARACTERIZATION OF PATTERNS OF FOREST FRAGMENTATION IN THE SOUTHWEST AMAZON: SATELLITE DATA ANALYSIS FROM 1986 TO 2005

Summary

While overall deforestation levels have been featured in discussions of the design and implementation of payments for ecosystem services (PES) programs, the landscape ecology literature has made clear that the spatial pattern of forest fragmentation is also crucial for forest ecosystem services such as carbon storage. Quantifying forest cover as well as forest fragmentation is thus very important for policy makers in order to appraise the value of environmental services, avoid forest loss, and compensate land users for environmental restoration. This paper combines land-cover analysis with satellite images and landscape ecology theory and methods in order to better understand patterns of landscape fragmentation for the purpose of environmental policy pertaining to PES. This study takes up the case of the Brazilian State of Acre in the southwestern Amazon. Acre has been an innovator concerning forest management policy and the state government has recently created a climate institute that houses a PES-carbon program. I focus on an analysis of forest cover and fragmentation using Landsat images for four Directed Settlement Projects (PADs) in Acre. Results from a comparison of time series imagery over roughly 20 years show that forest fragmentation in PADs has increased, resulting in smaller and more isolated forest fragments. Given that more isolated forest fragments tend to have lower productivity and thus ecological value, this is cause for concern among both policymakers and landowners. Monitoring of forest area and fragmentation patterns can thus inform both PES program implementation and land use practices.

Introduction

One of the main concerns associated with tropical deforestation is forest fragmentation. As fragmentation proceeds, habitat in patch size decreases, and patches become increasingly isolated in landscape mosaics with other types of vegetation (Laurence and Bierregaard 1997; Laurance, et al. 2002). Insofar habitats must be intact in order to provide ecosystem services, fragmentation undermines such services; and insofar habitats are highly biodiverse, fragmentation undermines species interactions and leads to biodiversity loss. The details of these ecological impacts depend on the specific spatial pattern of habitat fragmentation. In general, landscape ecology has shown that more intact landscape, larger habitat fragments, and fragments with strong connections (whether direct physical connections or smaller intervening distance) exhibit smaller ecological impacts.

Parallel to the literature on habitat fragmentation is the "road ecology" literature, which has similarly highlighted the importance of road networks as they divide habitats (Trombulak and Frissell 2000; Forman, et al. 2003; Coffin 2007). Road ecology can be thought of as a specialty within landscape ecology. Just as the spatial characteristics of habitat fragmentation help determine the ecological consequences, the spatial architecture of road networks in landscapes help determines the extent and intensity of ecological perturbations in landscapes. In general, forests fall and habitats are most perturbed along roads where accessibility is greater; consequently, ecological effects decline with distance from the roadside. This raises important questions about the design of roads networks in land settlements as it may greatly influence the spatial pattern of forest loss and thus the degree of ecological perturbation.

The spatial pattern of habitat fragmentation carries important implications for initiatives to prevent ecological degradation and /or encourage habitat restoration. One prominent example concerns payments for ecosystem services (PES) programs. The details of specific initiatives

vary, but the general logic is to price or otherwise value ecosystem services in order to generate incentives for landholders to engage in habitat conservation as a means of retaining such services. However, PES programs are often conceived in terms of aggregate measurement of habitat loss rather than accounting for fragmentation. For a given land area with a given amount of valuable habitat, greater fragmentation is likely to imply greater loss of ecosystem services, with financial implications for PES payments. Ecological restoration seeks to restore ecosystem services via reintroduction or support of native species. Restoration efforts rely on insights from landscape ecology closely tied to understanding the spatial pattern of habitat fragments, as by identifying key missing corridors to improve habitat connectivity. Greater fragment isolation usually implies greater costs at restoration efforts. Both PES and restoration initiatives thus rely on information about habitat fragmentation.

However, it is not enough to know what landscape's fragmentation pattern is at a given time; a full understanding of fragmentation also requires information about fragmentation dynamics. For example, if habitat loss involves reduction in the size of large blocks, that carries a worrisome set of implications for fragmentation and restoration, since large fragments are ecologically the most valuable. But if on the other hand the dynamic of fragmentation spares large blocks and instead eliminates already isolated fragments, maintenance of ecosystem services may be less difficult and costly. Hence it is crucial to understand not only the spatial pattern but also temporal dynamics of habitat fragmentation.

The Amazon is well-known for deforestation due to new land settlements for agriculture and ranching. Such settlements have varying road networks and land use patterns, resulting in diverse spatial patterns of forest fragmentation across the basin. Further, different highway corridors have opened at different times, and been populated by rural land settlements with contrasting road network designs as well as use rules concerning forest clearing. The result is

that roadside land settlements in the Amazon have yielded varying spatial as well as temporal patterns of forest loss and fragmentation. Given the considerable biodiversity and ecosystem services in the Amazonian forests, an analysis of the spatial patterns and temporal dynamics of forest fragmentation becomes important for implications for PES programs and other conservation efforts.

This chapter specifically focuses on the case of Directed Settlement Projects (PADs), which are among the oldest agricultural settlement projects category in the Brazilian Amazon. PADs however have been designed differently with regard to their road networks. Some PADs have the well-known "fishbone" road architecture, which entrains a forest fragmentation pattern involving long, thin strips of forest remnants in between parallel secondary roads. Other PADs however have roughly circular settlement patterns centered on a central hub from which secondary roads radiate out in different compass directions. Yet other PADs have still different road networks.

I therefore take up the question of whether one road network design within the PAD settlement type yields distinct levels of forest fragmentation from other types of networks. I compare four PADs in the eastern portion of Brazilian state of Acre. PADs there have distinct road networks and have incurred forest loss fragmentation over time. I evaluate the spatial patterns as well as temporal dynamics of forest fragmentation using several fragmentation pattern metrics in order to compare the PAD's. The findings show differences in spatial-temporal dynamics in fragmentation, which yield implications for conservation initiatives such as PES and restoration via improvements in fragments connectivity.

Deforestation, Forest Fragmentation and its Implications

Global deforestation has been heavily studied by the scientific community, with particular attention to the loss of tropical forests such as in the Amazon (Keller et al. 2009).

Tropical deforestation and forest degradation have received great attention in the last decade due to the implications for the world's atmospheric carbon (Rayner and Malone 2001). Recently, it has been estimated that 20% of carbon emissions came from tropical deforestation (Gullison et al. 2007; Boucher 2008). According to IPCC (2007), the burning of fossil fuels and clearing of forests have increased atmospheric carbon, contributing to global climate change. This issue has therefore increased international interest in approaches to reduce emissions from deforestation and forest degradation in developing countries (Metz et al. 2007; Ramankutty et al. 2007).

Further, deforestation results in forest fragmentation. With deforestation, landscapes become fragmented, resulting in a mosaic of patches of successional forest and agricultural lands. (Turner 2004; Turner 2001). Forest fragmentation is a major issue because fragmentation can isolate habitat patches, which entrains a series of negative ecological consequences (Laurance and Bierregaard 1997; Laurance et al. 2002; Mesquita et al. 1999; Forman 1997, Bierregaard 1992; Forman 1996). It can also change the ecological processes, such as nutrient cycling and pollination (Didham et al. 1996). Fragmentation can lead to plant mortality and cause biomass collapse in the forest. The rise in tree mortality alters canopy-gap dynamics (Ferreira and Laurance 1997; Laurance et al. 1998), which can further influence forest structure (Brokaw 1985; Hubbell and Foster 1986; Denslow 1987). These changes in turn lead to greater light penetration through a more open canopy, and increase forest litter, which can serve as fuel for fires. Tropical forest fragments are also drier than intact forest, making fragments vulnerable to fires during droughts (Nepstad et al. 2001; Laurance et al. 2001a). When necromass decomposes and fires spread, forest fragments emit carbon. In addition, fragmentation can influence species population (Laurance et al. 2001a) and alter the structure and dynamics of ecological communities (Laurance 2000; Laurance et al. 2001a). More specifically, fragmentation can reduce species richness and foster exotic species invasions.

Fragmentation is an eminently spatial process, and fosters the entry of external disturbances into the interiors of forest patches. While different disturbances penetrate to varying distances in fragments, fragment edges are the most impacted, highlighted thruogh the concept of "edge effects." The majority edge effects extend no further than 1 km (Murcia 1995), however some edge effects can extend as far as 5–10 km into the forest (Curran et al. 1999). Plant mortality rates are higher at forest edges than in remnant interiors (Laurance et al. 2001a).

On a larger scale, the size and shape and isolation of forest fragments greatly affect the extent of ecological degradation (Laurance and Bierregaard 1997; Laurance, et al. 2002). Larger fragments, fragments with more circular shapes and more area farther from edges, and fragments closer to other fragments exhibit less severe ecological degradation. In this context, the spatial organization of road networks helps define the geometry of forest fragments (Trombulak and Frissell 2000; Forman et al. 2003; Coffin 2007). The expansion of local road networks in colonization projects thus fragments forest landscapes into habitat mosaics with fragments defined by the spatial organization of roads in the projects. This raises questions about the design of colonization projects as it influences road network organization and entrains specific consequences for forest fragmentation.

PADs in the Brazilian Amazon

In the Brazilian Amazon, Direct Settlement Projects (PADs), are among the oldest colonization projects. PADs were established in the Amazon following Brazil's national strategy of regional occupation and development in its northern frontier in the 1970s. PADs were implemented with little consideration for landscape characteristics or environmental impacts of forest loss. Road networks in PADs originally followed the famous "fishbone" structure, defined by a primary road from which run perpendicular secondary roads parallel to each other (e.g., Moran 1981; Smith 1982; Fearnside 1986). The fishbone arrangement was adopted to maximize

access to land for productive use, but the straight lines for roads on the planning maps took no account of terrain or rivers (Perz, et al. 2008; Walker et al. 2011).

The focus on settlement for agricultural production disregarded deforestation and forest fragmentation as potential problems (Perz, et al. 2008). Consequently, deforestation in PADs primarily occurs along the roadsides, and follows the fishbone pattern. Previous work on deforestation in settlement projects shows that the "fishbone" pattern of fragmentation faithfully follows the road network (Arima et al. 2005, 2008; Perz et al. 2008). Over time, as secondary roads in PADs have been extended, the fishbone pattern has also expanded (Perz et al. 2007), incrementing forest fragmentation into increasingly long, thin strips of remnant forest cover and thereby reinforcing the fishbone fragmentation geometry (Perz et al. 2008).

That said, different PADs have been designed around distinct road network designs (Perz, et al. 2008; Batistella et al. 2003); the fishbone pattern is only the best-known. This carries the implication that the spatial pattern of forest loss and thus fragmentation will also vary among PADs with different designs. This raises important questions about the design of different PADs and the implications for fragmentation patterns.

The Study Area and Research Design

In this context, the Brazilian State of Acre is an interesting study case because it includes multiple PADs. In particular, there are several PADs in eastern Acre which are similar in several respects – they are located near to each other and were established at similar times – but which have very different road networks. This provides a useful research design for case comparisons of the spatial patterns and temporal dynamics of forest fragmentation in land settlements that are comparable in several respects but differ in terms of a key factor influencing fragmentation dynamics.

Further, the government of Acre has recently created a state-level Climate Change Institute that houses a PES program, notably including a carbon PES initiative (Alencar et al. 2012; Governo do Acre 2009). The carbon PES program will seek to benefit landholders who conserve carbon in standing forest biomass. Notably, the carbon PES program seeks to evaluate environmental services of forest fragments as a means of compensating landholders for restoration of degraded areas. The program is relevant for present purposes because it will target PADs in eastern Acre precisely because they have degraded land. Insofar as forest fragmentation proceeds alongside forest degradation due to fragment isolation and decreased patch size, the somewhat unique emphasis on forest restoration in Acre's carbon PES is eminently relevant to the study of forest fragmentation. While the carbon PES program has not yet been implemented, an analysis of forest fragmentation provides a useful input for understanding prospects for ecosystem services restoration. For example, highly isolated fragments are likely to require greater investments to be linked to other fragments. But first, there is a need for a systematic analysis of fragmentation patterns and dynamics, especially as they occur across land settlements with different types of road networks.

The research area encompasses four PADs, located in a recently created zone in southeastern Acre called "priority attention zones" (PAZs) which is part of the special development zones (EDZs) of the state government (Figure 4-1). PAZs and EDZs are new development zones created by the Ecological Economic Zoning plan (ZEE) (Governo do Acre 2006). These areas are priority targets for Acre's new public policies that seek to reconcile economic growth with environmental sustainability (Governo do Acre 2006). Notably, eastern Acre encompasses PADs targeted by Acre's Climate Change Institute and its carbon PES program. The prospective implementation of the carbon PES program may not only affect fragmentation in these PADs. In addition, the historical pattern of the emergence of

fragmentation in these PADs bears ramification for the viability and attractiveness of adoption of the carbon PES program. In that context, eastern Acre is a useful study case for an evaluation of forest fragmentation because PADs in eastern Acre are of differing sizes and have distinct road networks. The consequence is that fragmentation patterns in PADs in Acre will also differ.

A final reason to focus on eastern Acre is because the PADs also vary in terms of their locations. The PADs are located along the newly paved highway, BR-317, also known as the Inter-Oceanic Highway, or IOH (CEPEI 2002; Killeen 2007). Paving of the IOH proceeded under several Brazilian infrastructure plans (Perz et al. 2010; Mendonza et al. 2007) and the Initiative for the Integration of Regional Infrastructure in South America (IIRSA). Both prioritize economic development via regional integration through infrastructure upgrades. This is significant for the development contemplated implies deforestation for agricultural production for export, and runs contrary to the goals of the carbon PES program, which seeks to raise incomes via forest conservation. However, some PADs are closer to Rio Branco, the capital of Acre, where deforestation is greater. Hence the choice of PADs in eastern Acre permits an evaluation of forest fragmentation under distinct circumstances tied to settlement design and location, as well as comparisons in light of very different public policies.

Data and Methods

Evaluation of forest fragmentation requires accurate geographic information of land-cover over time (Lambim 1999; McGuire et al. 2001). Remote sensing (RS) accurately obtains information of land cover conversion rapidly, and over a range of spatial and temporal scales. In particular, satellite RS data allow for standardized observations of land cover with considerable spatial detail. Satellite RS therefore affords the opportunity to evaluate forest fragmentation with comparable data for different locations and over time. In this chapter, I observe land cover trajectories over roughly 20 years in approximately 5-year intervals. Specifically, data come

from Landsat imagery for eastern Acre in 1986, 1991, 1996, 2000 and 2005. This permits a time series analysis to evaluate deforestation and forest fragmentation since the period when PADs were created in eastern Acre thru the completion of paving the IOH in 2002.

I focus on four PADs in eastern Acre: PAD Humaitá, PAD Peixoto, PAD Quixadá and PAD Quixadá Gleba 6. Selection of these four PADs permits analytical comparisons to evaluate the importance of proximity to Acre's capital of Rio Branco, where deforestation is greater than elsewhere in Acre, as well as road network design, as some PADs follow the classic fishbone network whereas others do not. With regard to distance from Rio Branco, two PADs are close to the city while the other two are more distant. Whereas PAD Peixoto and PAD Humaitá are located close to Rio Branco (just to the north and east of the city; Figure 4.1), PAD Quixadá and PAD Quixadá Gleba 6 are located at the other end of the IOH in Acre, near the tri-national frontier with Peru and Bolivia. This permits comparisons between the "high" deforestation PADs close to Rio Branco – Peixoto and Humaitá – with the "low" deforestation PADs, namely Quixadá and Quixadá Gleba 6. I anticipate that fragmentation will be greater in the PADs close to Rio Branco than elsewhere.

However, road network design is also likely to affect relative levels of fragmentation. I therefore also selected these four PADs to permit comparisons within the two locations in terms of road network design (Figure 4.2) Among the PADs close to Rio Branco, Peixoto exhibits the traditional fishbone structure of secondary roads running perpendicularly from the highway, whereas Humaitá exhibits a radial road network structure emanating from a hub in the center of the PAD. Similarly, among the PADs far from Rio Branco, Quixadá also exhibits hallmarks of the fishbone structure as evident in parallel secondary roads, whereas Quixadá Gleba 6 runs along the IOH, without many road intersections. If deforestation proceeds along road frontage in lots within these PADs, we should expect distinct patterns of forest fragmentation geometry.

To evaluate forest fragmentation in these PADs, I acquired satellite imagery available for the Amazon's annual dry-season, specifically from the months of July to November. As discussed elsewhere, I pursued radiometric calibration, geometric registration (image to image rectification), normalization for precipitation differences (when necessary), and mosaicking. Pre-processing is very important not only to allow comparisons among PADs and over time, but also to permit better visual quality of the data which then facilitates more reliable image classification.

The transformation of spectral data into earth surface information through the extraction of thematic features (forest and non-forest land cover) has been traditionally done through classification techniques. Since my analysis focuses on forest and non-forest cover, clouds, shadows and water from each image were removed before classification. Afterwards, it was applied a Principal Component Analysis (PCA) and image differencing and thresholding method (Varlyguin et al. 2001). PCA of satellite images is widely used to extract useful information from multiple bands by filtering noise from the data. A tasseled cap index (Kauth et al. 1976), a mid-infrared index (Boyd and Petitcolin 2004), and a 3-by-3 moving window calculation of the variance of each pixel for bands 4, 5 and 7(the near and mid-infrared bands) was performed as secondary product to help with image classification. Due to striping in the visible and thermal bands, a rule-based or decision tree classification was applied (Breiman 1984) instead of traditional unsupervised or supervised techniques. The rule-based classification provided flexibility to eliminate visible and thermal bands, using only the near-and mid-infrared bands along with secondary derived products. I also conducted field work in Acre to help classify land cover and improve classification accuracy. Field visits involved collection of training samples for accuracy assessment of forest-non-forest (F-NF) classifications. I initially classified land

cover into forest, pasture and bare-built, and then aggregated pasture and bare-built to create the non-forest class.

I then subset the four PADs out of the classified TM and ETM+ images for each observation year, yielding a series of land cover maps for each PAD (Figure 4-3, 4-4, 4-5 and 4-6). I transformed the PAD land cover maps to grids for analysis of forest fragmentation. The PAD grid maps then served as data for the fragmentation software FRAGSTATS, which computes landscape fragmentation metrics.

Fragmentation analysis involves a variety of pattern metrics which describe different aspects of patch isolation and shape (Southworth et al. 2004). These metrics quantify specific spatial characteristics of land cover patches in landscapes.

I generated pattern metrics at both the landscape and class level. According to Herzog (2002), landscape metrics describe the composition and configuration of the overall landscape of a study area. By contrast, class metrics capture patterns in all patches of a given land cover class (Yu and Ng 2006). Hence whereas landscape metrics characterize fragmentation in a landscape mosaic, class metrics focus on fragmentation in a given land covers class. In this study, given the importance of ecological consequences of forest fragmentation, I focus on class metrics for forest and non-forest cover.

Although there is no 'best' choice for which metric describes a particular pattern, I selected specific pattern metrics based on a review of landscape fragmentation analysis (Chavez 2009; Nagendra et al. 2004 and Southworth et al. 2002). I consulted the FRAGSTATS website to select pattern metrics for this study (www.umass.edu/landeco/research/fragstats.html). Tables 4-1 and 4-2 summarize the pattern metrics I selected for this study. Table 4-1 outlines six landscape pattern metrics: largest patch index (LPI), edge density (ED), mean patch area (MP-AREA), number of patches (NP), patch density (PD), contagion (CONTAG), and the Shannon

Diversity Index (SHDI). In general, values for these indexes change as landscapes become fragmented into mosaics of different land covers. In the study area, as deforestation proceeds, insofar as forest patches become fragmented into irregular shapes and isolated, and insofar as patches of other land covers emerge in the landscape, the largest patch size declines, edge density rises, mean patch size drops, the number of patches increases, contagion is reduced, and the Shannon Diversity Index grows.

In addition, as landscapes become fragmented, it is often important to know about fragmentation of particular land cover classes. In the case of the Amazon and other forested regions, there is particular interest in knowing about forest fragmentation. I therefore calculated class pattern metrics for forest and non-forest cover. Table 4-2 outlines a suite of class pattern metrics: percentage of the landscape in that cover class (PLAND), largest patch index for the class (LPI), total edge for the class (TE), edge density for the class (ED), mean patch area (MP-AREA), number of patches in the class (NP), patch density for the class (PD), and class patch cohesion (COHESION). As forest fragmentation proceeds, values for these pattern metrics will change, but differently for forest and non-forest cover classes. As fragmentation increases, forest class pattern metrics will change as follows: percentage of forest declines, largest forest patch drops, total edge along forest patches rises, edge density in forest patches increases, mean forest patch area declines, number of forest patches rises, forest patch density increases, and forest patch cohesion (connectedness) is reduced. Conversely, as fragmentation proceeds, non-forest pattern metrics will do just the opposite, from a rise in percentage non-forest to a rise in non-forest cohesion.

The analysis involved calculation of each pattern metric for every time point (from 1986 to 2005) for each of the four PADs. This permits an evaluation of changes in different landscape

and class pattern metrics for fragmentation over time as well as comparisons of fragmentation among the PADs in terms of distance from Rio Branco and road network design.

Results and Discussion

Land Cover Analysis

The first part of my analysis focuses on basic indicators of land cover change and interpretations of fragmentation patterns using maps. Figures 4-3 and 4-4 show land cover change trajectories in the four PADs from 1986 to 2005. Figure 4-3, for percent forest cover, makes evident that forest has been cleared over time in all four PADs. Figure 4-4, for percent non-forest cover, shows that deforestation was low in all four PADs as of 1986, but rose substantially thereafter. In particular, deforestation is relatively high in PAD Peixoto and Humaitá, the PADs near Rio Branco where I anticipated greater forest clearing due to proximity to the city.

The maps presented in Figures 4-5 to 4-8 not only indicate rising deforestation as noted above, but also show increasing isolation of forest patches over time. Also evident in each of these figures are the road networks of each PAD, along which deforestation has begun and then expanded. The fishbone networks in PAD Peixoto and Quixadá are also evident in Figures 4-5 and 4-7, respectively; the radial design in PAD Humaitá is evident in Figure 4-6; and the elongated road network in PAD Quixadá Gleba 6 appears in Figure 4-8. These four figures confirm that deforestation proceeded primarily along roads, and that fragmentation patterns reflect road networks.

Landscape Metrics

The second part of the analysis goes beyond the thematic maps discussed above to evaluate forest fragmentation using the landscape and class pattern metrics. Figures 4-9 thru 4-30 present the landscape pattern metrics for the four PADs from 1986 to 2005. Each figure permits

comparisons among the four PADs over time for a given landscape metric. This permits a comparative analysis of landscape fragmentation among the four PADs, in order to evaluate their fragmentation trajectories and the importance of distance to Rio Branco and PAD road network design for fragmentation.

Figure 4-9 compares the largest patch index (LPI) for the landscapes in the four PADs. Overall, LPI declines substantially from 1986 to 2005, which indicates considerable fragmentation over time. However, the trajectories differ among the PADs and there are differences evident by 2005. A road network effect appears in 2005, such that LPIs in the "fishbone" PADs, Peixoto and Quixadá, are lower than elsewhere. This may be a temporary state of affairs however, for LPIs values vary substantially among time periods.

Figure 4-10 presents edge densities (EDs) for the four PADs. EDs rise substantially over time, and there is some differentiation by 2005. There is some evidence of a distance effect: EDs are lower in the PADs closer to Rio Branco, Peixoto and Humaitá, than those far from the capital city. In this sense, there is greater landscape fragmentation closer to the city.

Figure 4-11 compares mean patch areas (MP-AREA) in the PADs. MP-AREA declines substantially to roughly 7 ha in 1996 and then levels off, and there are limited differences by distance or road network structure among the PADs. PADs with greater MP-AREA values change from one time point to the next.

Figure 4-12 shows findings for the number of patches (NP) in the four PADs. Here a large scale effect appears, for Peixoto is considerably bigger than the other areas and thus not surprisingly has more patches. Interestingly, Humaitá is larger than Quixadá but has a similar number of patches, which could be taken to indicate that Quixadá is more fragmented. To that extent, one could interpret Figure 4-12 as suggesting greater fragmentation in the PADs according with scale effect.

Figure 4-13 compares contagion values for the four PADs. Landscape contagion declines over time in all four PADs, a reflection of increasing fragmentation and patch isolation. Further, contagion values do not differ substantially among the PADs.

Figure 4-14 evaluates patch densities among the four PADs. Patch densities increase overall, another indication of increasing landscape fragmentation, and trajectories differ among the four PADs. The trajectories in the "fishbone" PADs, Peixoto and Quixadá, are similar, though Peixoto exhibits a lower patch density than Quixadá. These findings could be taken to suggest similarities in fragmentation due to similar road networks, moderated by differences stemming from contrasting distances to Rio Branco. In addition, Humaitá has a relatively low patch density like Peixoto, the other PAD near Rio Branco, and Quixadá Gleba 6 has a relatively high patch density like Quixadá. Hence one might argue for at least weak road network and distance effects on patch densities in the PADs.

Figure 4-15 concludes the landscape fragmentation analysis by comparing the Shannon Diversity Index (SHDI) in the four PADs. In general the SHDI rises over time, a further indication of growing landscape fragmentation. Moreover, there are few differences among the PADs, and the differences become smaller still as time passes.

Overall, the landscape pattern metrics indicate that 1) landscape fragmentation has risen over time in the four PADs, but that 2) differences only appear for certain pattern metrics, and 3) while there is evidence of both distance and road network design effects, neither consistently explains variation in differences among landscape fragmentation among PADs. One might interpret these findings as implying that while PADs beget landscape fragmentation, there are not major differences in fragmentation among PADs that ought to preoccupy policymakers interested in carbon PES programs.

That said, landscape fragmentation indexes can obscure class-specific fragmentation patterns. The last part of the analysis therefore presents findings for forest and non-forest pattern metrics. I present results by indicator, with a joint presentation of findings for a given class pattern metric for both forest and non-forest.

Figures 4-16 and 4-17 show that the percentage of landscapes in forest decreases while the percentage of non-forest increases in the PADs, respectively. These figures confirm earlier findings of forest decline and the expansion of deforestation. They also confirm the distance effect noted earlier, in that forest decline and the corresponding rise in deforested area are more rapid in Peixoto and Humaitá, located closer to Rio Branco.

Figures 4-18 and 4-19 present findings for the largest patch index (LPI) for forest and non-forest cover. Not surprisingly, the size of the largest forest patches declines while the largest non-forest patches rise over time. However, differences appear among PADs for both forest and non-forest LPIs. Forest LPI trajectories differ substantially among the PADs, and by 2005, a weak distance effect appears, such that forest LPIs are smaller in the PADs close to Rio Branco. Non-forest LPIs however tell a somewhat different story, in that road network design appears more important. The two fishbone PADs, Peixoto and Quixadá, have similar non-forest LPI values by 2005, with the radial PAD, Humaitá, with a substantially larger value and the elongated PAD, Quixadá Gleba 6, with a lower value.

Figures 4-20 and 4-21 show results for total edges for forest and non-forest cover in the four PADs. Here the findings are very similar, as both forest and non-forest edges are dominated by scale effects, such that Peixoto exhibits greater edge lengths than the other PADs and Quixadá Gleba 6 has the least. The findings for total edges do not provide strong evidence of either distance or road network effects on fragmentation.

Figures 4-22 and 4-23 evaluate that contention by comparing forest and non-forest edge densities, which are adjusted for land areas. Edge densities for forest and non-forest both rise over time, indication of increasing interspersing of forest and non-forest patches over time. Here a distance effect appears, in that PADs near Rio Branco, Peixoto and Humaitá, exhibit somewhat lower edge densities. This is a somewhat surprising finding insofar as deforestation is more extensive in those PADs and higher edge densities suggest greater fragmentation. One interpretation is that deforestation is beginning to dominate the landscapes of PAD Peixoto and Humaitá, reducing edge densities.

Figures 4-24 and 4-25 permit analysis of mean forest and non-forest patch areas in the PADs. Mean forest patch sizes decline precipitously in all PADs, mirrored by the rise in mean non-forest patch areas in the PADs during the same time period. Whereas differences in mean forest patch areas largely converge (though mean patch sizes are slightly larger in Quixadá Gleba 6), there is a divergence in mean non-forest patch areas among the PADs. Here a substantial distance effect appears: mean non-forest patches become considerably larger in the PADs close to Rio Branco, Peixoto and Humaitá, but remain relatively small in the distant PADs, Quixadá and Quixadá Gleba 6.

Figures 4-26 and 4-27 compare the number of forest and non-forest patches among the PADs. As seen earlier, the number of patches differs among the PADs due to a scale effect, with many more patches in Peixoto due to its larger size. The elongated road network in Quixadá 6 does differentiate its forest patches from the other PADs, but the road network effect is not evident for non-forest patches.

Figures 4-28 and 4-29 consider patch densities to control for the scale effects seen in the two previous figures. Forest patch densities rise in all four PADs, evidence of forest fragmentation, and differences not clearly correspond to distance or road network designs. At

first glance, non-forest patch densities exhibit diverse trajectories, but careful inspection of Figure 4-29 reveals very similar trajectories for the fishbone PADs, Peixoto and Quixadá, though non-forest patch densities are consistently higher in Quixadá. In turn, the relatively low non-forest patch densities in Peixoto and Humaitá suggest a distance effect, with non-forest predominating near Rio Branco.

Figures 4-30 and 4-31 conclude the analysis of class pattern metrics by evaluating forest and non-forest cohesion (connectedness) among the PADs over time. There is an interesting contrast that summarizes previous findings: whereas forest cohesion declines over time, non-forest cohesion rises. This reflects increasing deforestation as well as forest fragmentation. In addition, there is a divergence in forest cohesion and a convergence in non-forest cohesion. With regard to forests, there is arguably a road network design effect in that the radial network structure in Humaitá exhibits lowers cohesion values. However, cohesion values do not vary much overall (98 to 99).

Conclusion

The analysis shows that the four PADs in eastern Acre experienced considerable deforestation and forest fragmentation from 1986 to 2005. Whereas the PADs were largely forested when created in the 1980s, they had lost half or more of their forest cover by 2005. In addition, landscape and class pattern metrics indicate considerable forest fragmentation in the PADs over the same period. As forest cover declined, forest patches became smaller, more numerous, more irregular, and more isolated (less connected). This reflects land cover conversion from forest to agriculture, following the intent of establishing the PADs for purposes of rural production.

The analysis featured a comparative analysis of fragmentation trajectories among PADs with different distances to the Acrian capital of Rio Branco and contrasting road network

structures. The results show that for some pattern metrics, fragmentation trajectories do vary among the PADs. However, for many pattern metrics, there are not large differences in fragmentation among the PADs. Some evidence of distance effects does appear: among the landscape pattern metrics, PADs closer to the city exhibit lower edge densities and lower patch densities. Similarly, there is limited evidence among landscape pattern metrics that road network design in PADs matters: the fishbone PADs had lower largest patch indexes. Similarly, there are some indications that distance effects are important for the class pattern metrics: in the PADs closer to the city, there is less forest, largest forest patches are smaller, forest and non-forest edge densities are lower, mean non-forest patch areas are larger, and non-forest patch densities are lower. And there is some evidence that road network design affects class fragmentation: in fishbone PADs, non-forest largest patches are similar.

Results infer that evidence of distance effects appear due to deforestation in the PADs near Rio Branco to be higher, about 60-70% by 2005, when compared to PADs located far from Rio Branco, where deforestation is only about 50%. When deforestation is roughly 50%, some pattern metrics take higher values since the landscape is more heterogeneous than when the landscape is more homogeneous, as when either forest or non-forest begin to dominate. In the near Rio Barnco PADs, results showed something like 30% forest and 70% non-forest, so the landscape is more homogeneous; on the other hand, far from Rio Branco PADs, the landscape is more 50/50, which is more heterogeneous. So we can expect more homogenous forested landscape with consequently smaller degree of fragmentation metric indices, smaller number of patches (NP), edge density (ED), and bigger for mean patch size (MPS), contagion (CONT), and Large patch index (LPI). While a more heterogeneous forested landscape will show a larger degree of fragmentation and the following expected metric indices: larger NP, ED, and a smaller

MPS, CON. Nevertheless, the outcome will be dependent on type of land (settlement category) at specific time periods; this will enhance or reduce spatial homogeneity and heterogeneity.

Overall, to the extent that there are differences in fragmentation among PADs, they appear to stem from the more extensive deforestation in PADs close to Rio Branco, which results in more non-forest, smaller forest patches, larger non-forest patches, and so forth. Hence there is more evidence that distance from the capital affects fragmentation than road network design. But differences are not generally large, and they often fluctuate over time in the trajectories, which implies that observed differences by 2005 may be temporary.

Observation of fragmentation trajectories opens the possibility of capturing changes in fragmentation differences among PADs. However, while some pattern metrics indicated very different trajectories among PADs over time, most did not. The one potential exception concerns the last period between the two most recent time points for which I observed fragmentation: during 2000-2005, deforestation rose faster than before, and some differences appeared in fragmentation among the PADs. An implication of this finding is that the timing of observations of fragmentation is potentially important, which affirms the importance of observing fragmentation trajectories and not just one point in time.

But given the extensive land cover change in the PADs, there is also reason to expect differences in fragmentation among PADs to remain limited into the future. Deforestation levels now reach or exceed those legally allowed in the PADs, making future deforestation increasingly problematic. If future deforestation is more limited, it is also likely that differences in fragmentation among the PADs will remain limited also.

The findings presented here and the prospects for limited future differences in fragmentation among PADs carry policy implications, as for carbon PES programs. If differences in fragmentation are limited, then carbon PES programs should broadly target PADs

rather than only PADs with specific road network designs. To the extent that carbon PES programs should target some types of PADs over others, they should target PADs farther from Rio Branco rather than PADs with specific types of road networks. The weak findings for road network design are somewhat surprising given the large differences in the road network designs among the PADs and previous work indicating that roads matter for habitat mosaics. An implication is that the neglect of landscape ecology in the design of PAD road networks does not by itself yield enduring problems for forest conservation, at least insofar as one road network design does not yield substantially worse fragmentation than another. But if the goal of carbon PES programs is to secure forest fragments in landscapes with greater patch size and connectivity, there is not a strong basis for targeting, and to the extent that there is targeting, it should focus on PADs farther from Rio Branco rather than PADs with specific types of road networks.

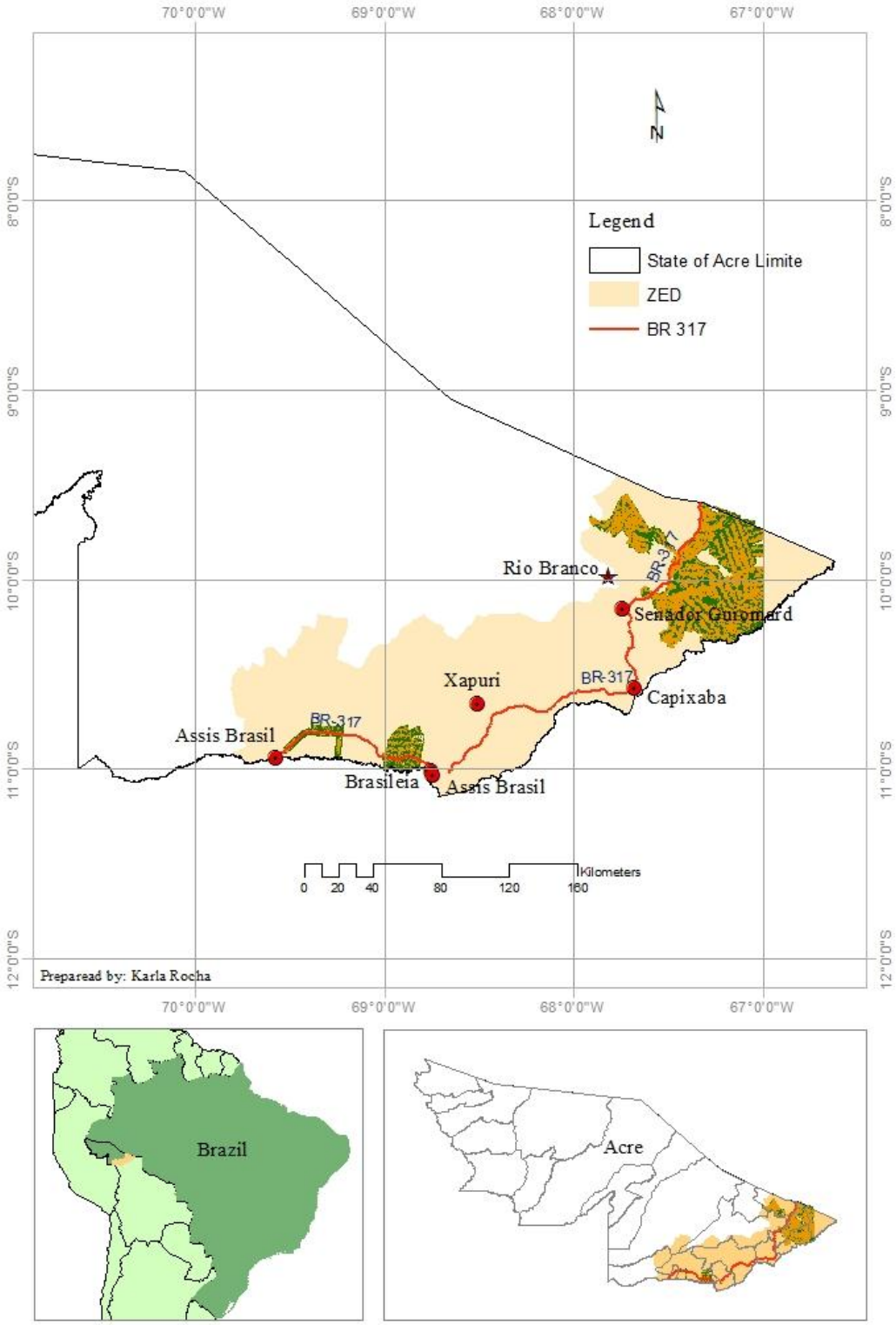

Figure 4-1. Research area in the southeast state of Acre, Brazil.

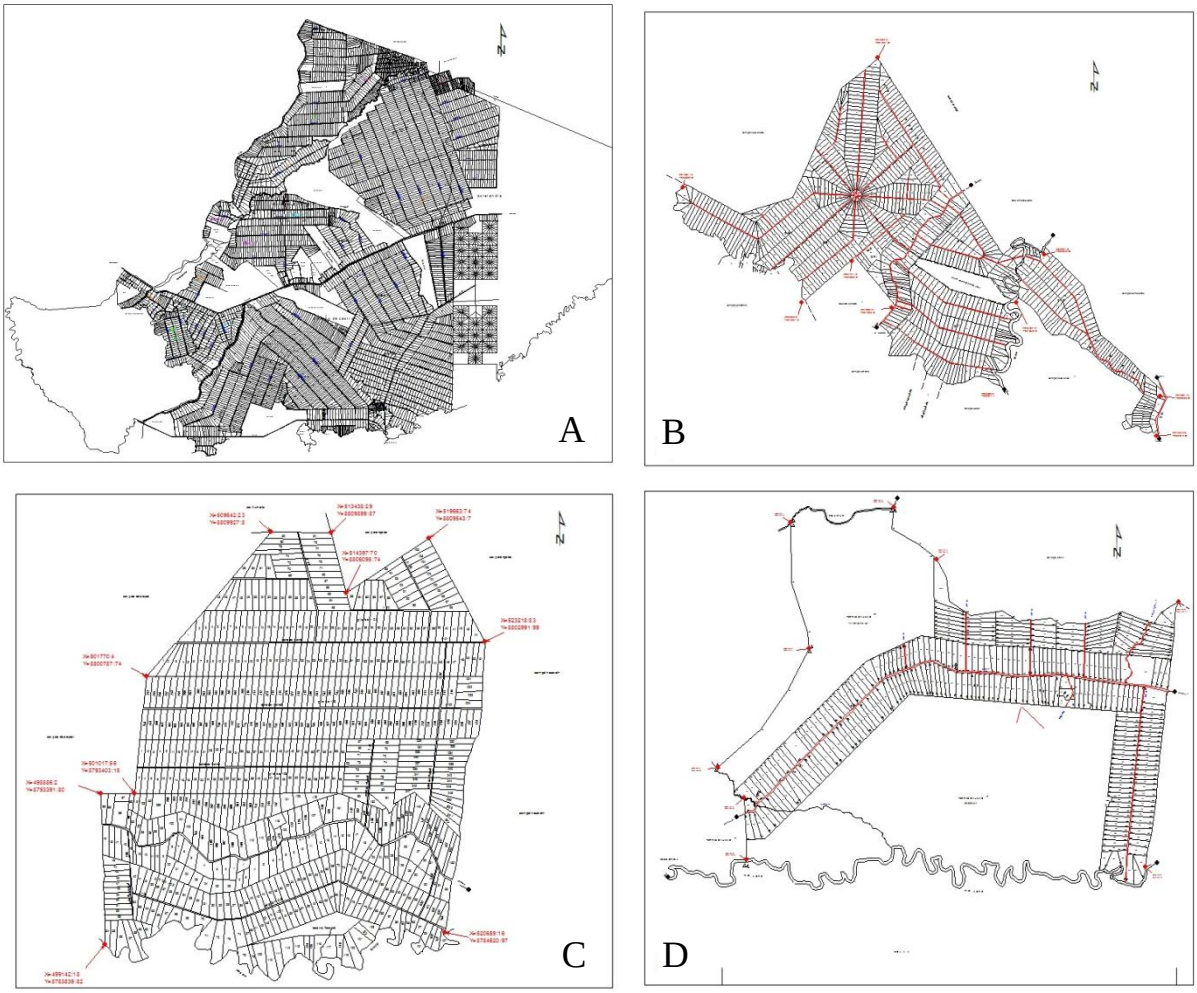

Figure 4-2. Road network design at PAD. A) Pedro Peixoto. B) Humaitá. C) Quixadá. D) Quixadá Gleba 6 (recentrly been coverted as part of Santa Quitéria settlement).

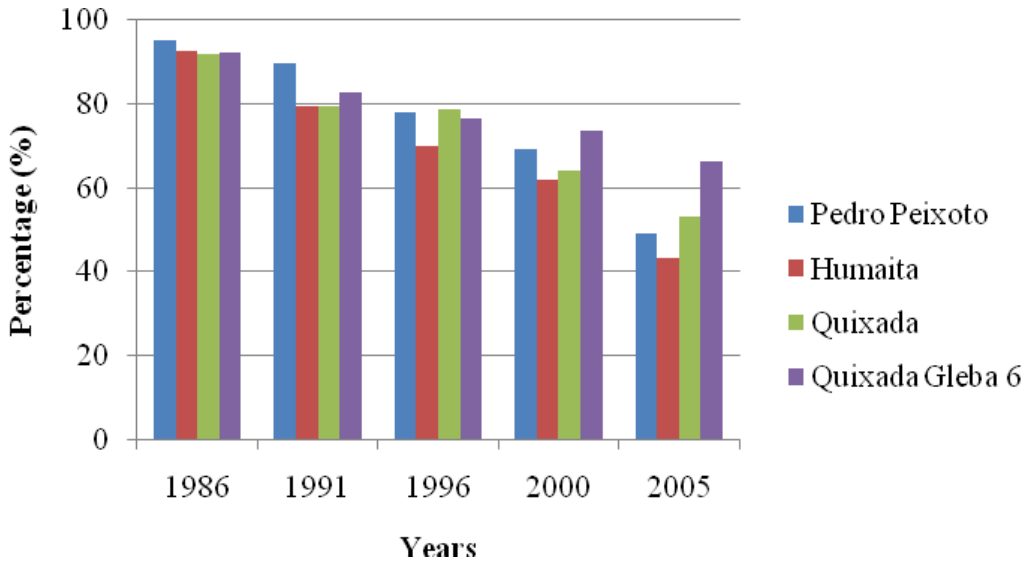

Figure 4-3. Percentage of forest in each specific time period.

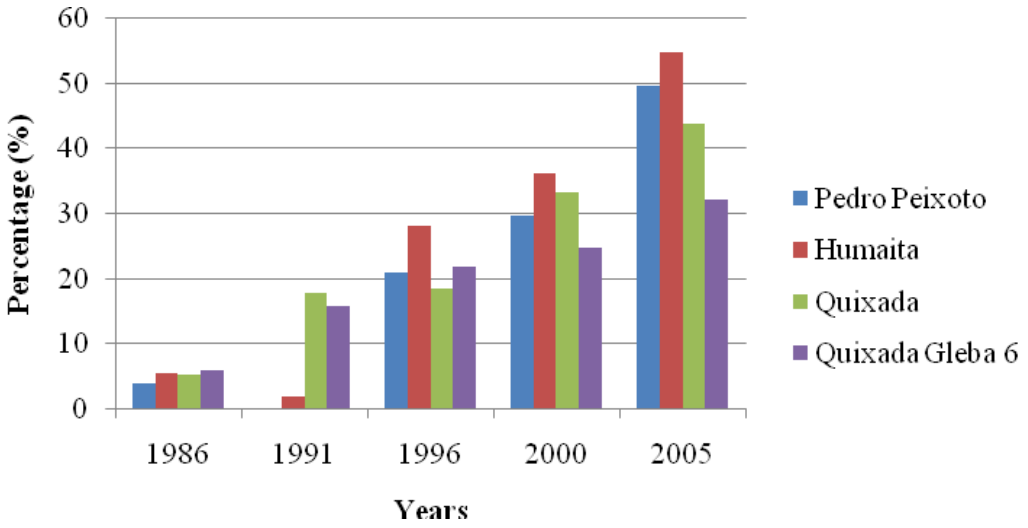

Figure 4-4. Percentage of non-forest in each specific time period.

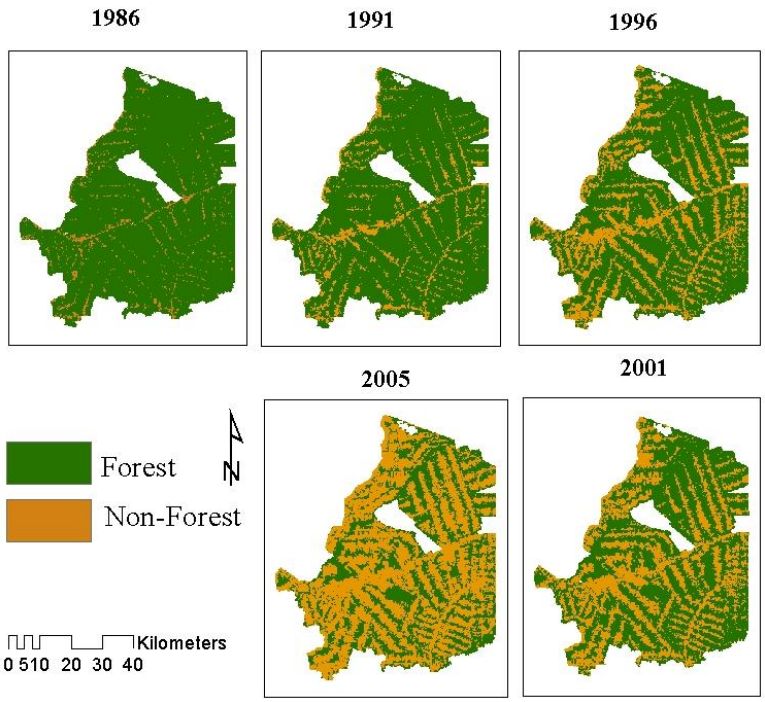

Figure 4-5. Land-cover change within PAD Pedro Peixoto from 1986 to 2005.

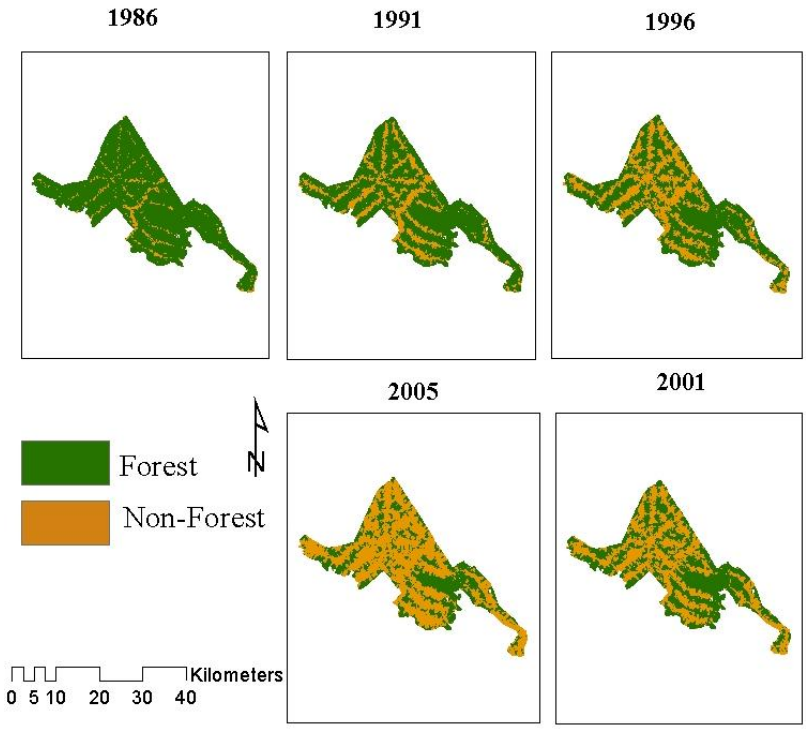

Figure 4-6. Land-cover change within PAD Humaitá from 1986 to 2005.

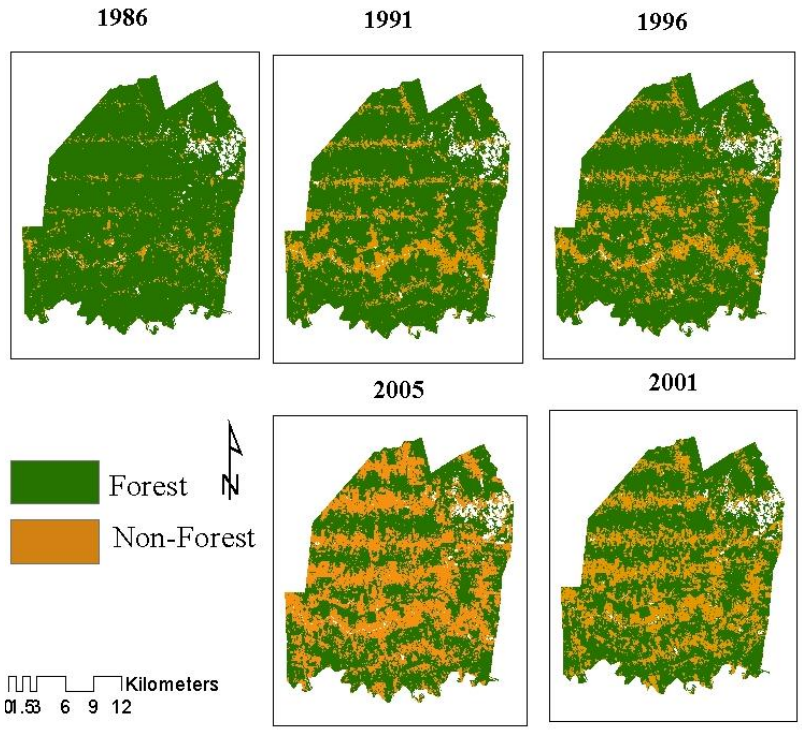

Figure 4-7. Land-cover change within PAD Quixadá from 1986 to 2005.

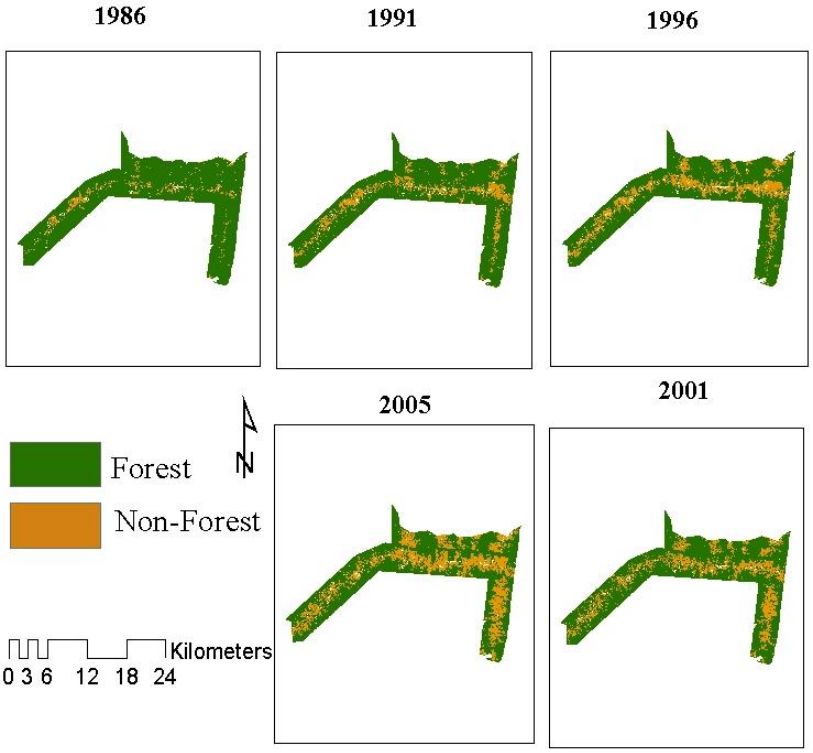

Figure 4-8. Land-cover change within PAD Quixadá Gleba 6 from 1986 to 2005.

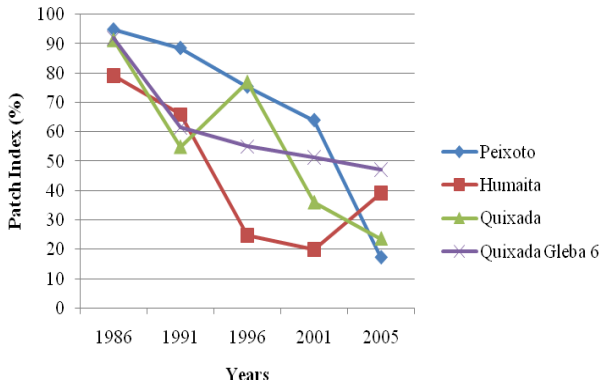

Figure 4-9. Largest patch index in landscapes in four PADs in eastern Acre - Brazil, 1986-2005.

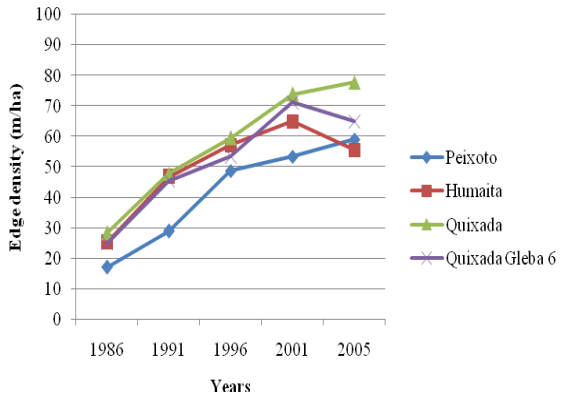

Figure 4-10. Edge densities in landscapes in four PADs in eastern Acre - Brazil, 1986-2005.

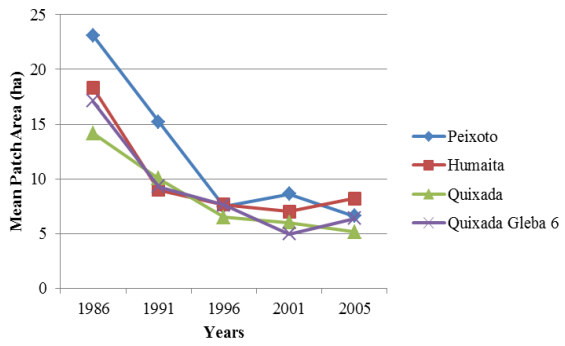

Figure 4-11. Mean patch areas in landscapes in four PADs in eastern Acre - Brazil, 1986-2005.

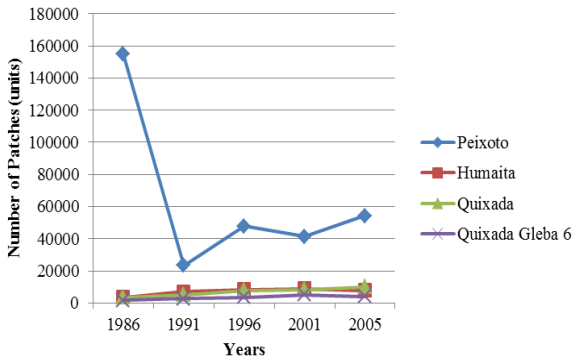

Figure 4-12. Number of patches in landscapes in four PADs in eastern Acre - Brazil, 1986-2005.

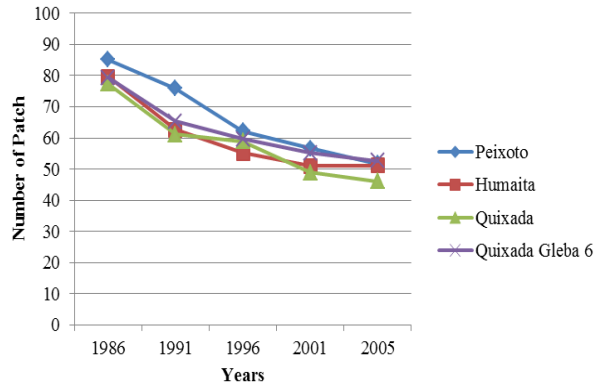

Figure 4-13. Contagion in landscapes in four PADs in eastern Acre - Brazil, 1986-2005.

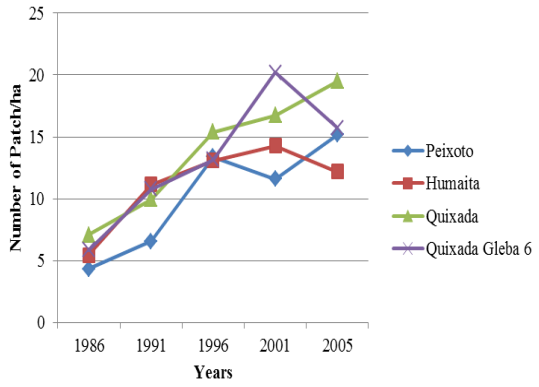

Figure 4-14. Patch density in landscapes in four PADs in eastern Acre - Brazil, 1986-2005.

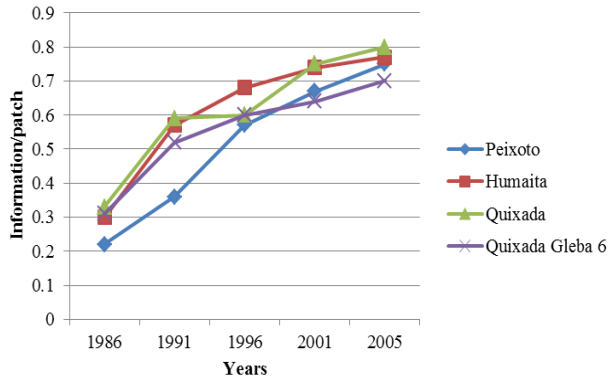

Figure 4-15. Shannon diversity index for landscapes in four PADs in eastern Acre - Brazil, 1986-2005.

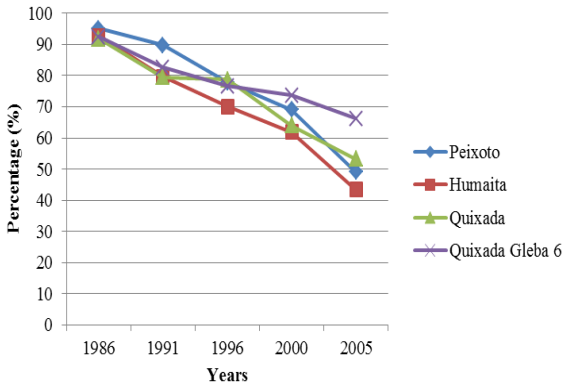

Figure 4-16. Percentage of landscape in forest in four PADs in eastern Acre - Brazil, 1986-2005.

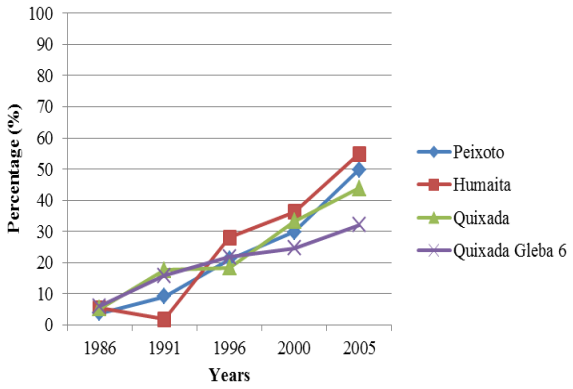

Figure 4-17. Percentage of landscape in non-forest in four PADs in eastern Acre - Brazil, 1986-2005.

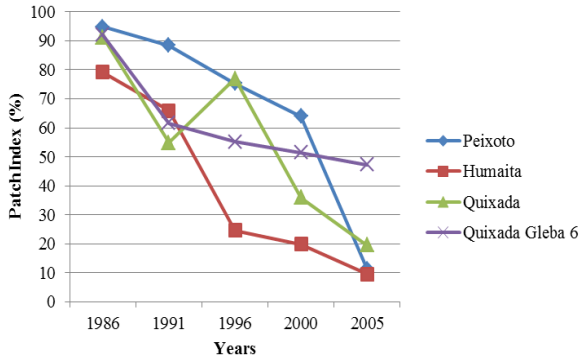

Figure 4-18. Largest patch index in forest in four PADs in eastern Acre - Brazil, 1986-2005.

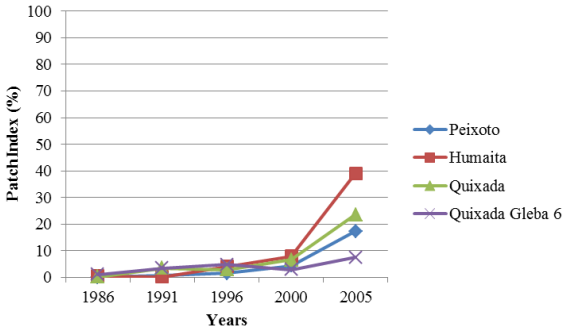

Figure 4-19. Largest patch index in non-forest in four PADs in eastern Acre - Brazil, 1986-2005.

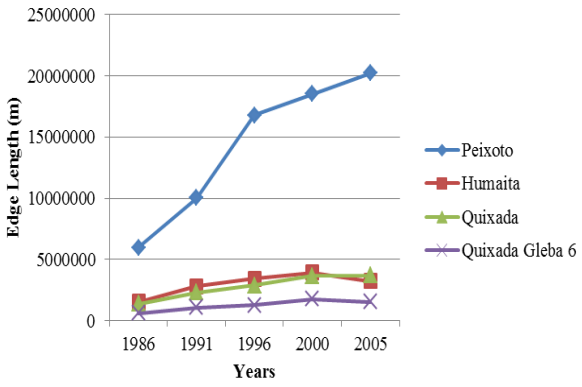

Figure 4-20. Total edge in forest in four PADs in eastern Acre - Brazil, 1986-2005.

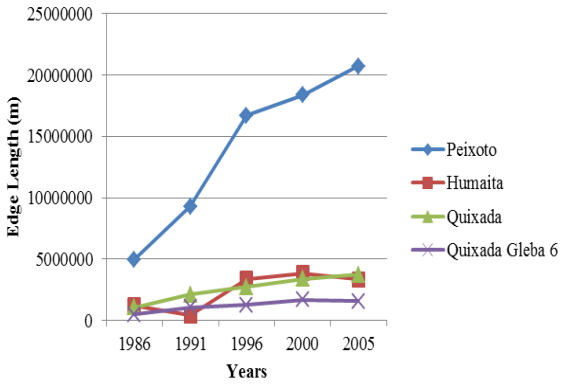

Figure 4-21. Total edge in non-forest in four PADs in eastern Acre - Brazil, 1986-2005.

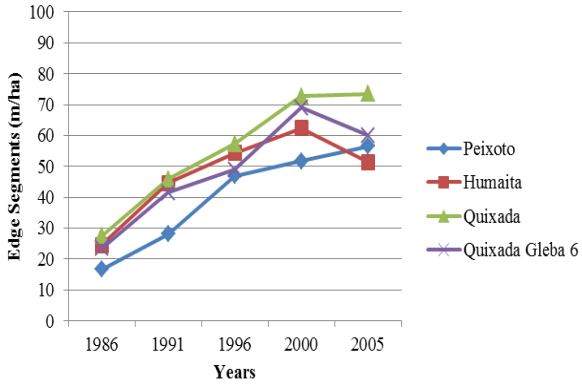

Figure 4-22. Edge density in forest in four PADs in eastern Acre - Brazil, 1986-2005.

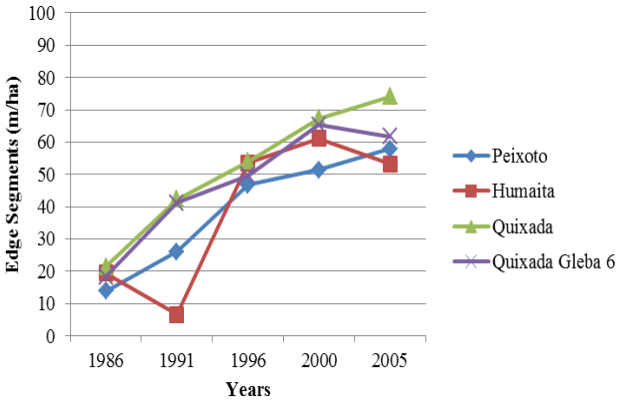

Figure 4-23. Edge density in forest in four PADs in eastern Acre - Brazil, 1986-2005.

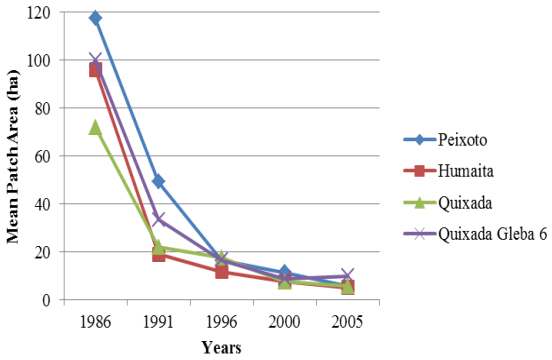

Figure 4-24. Mean patch area in forest in four PADs in eastern Acre - Brazil, 1986-2005.

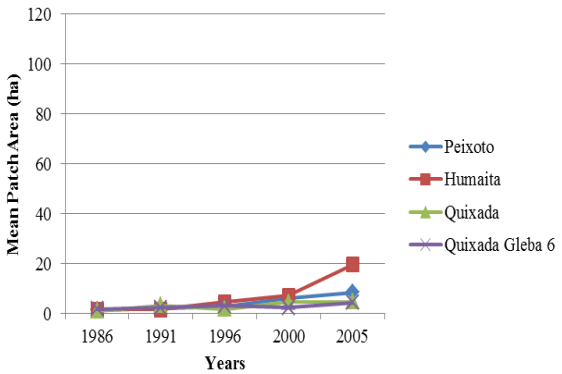

Figure 4-25. Mean patch area in non-forest in four PADs in eastern Acre - Brazil, 1986-2005.

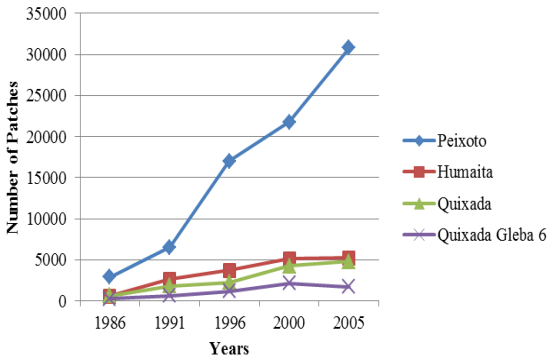

Figure 4-26. Number of patches in forest in four PADs in eastern Acre - Brazil, 1986-2005.

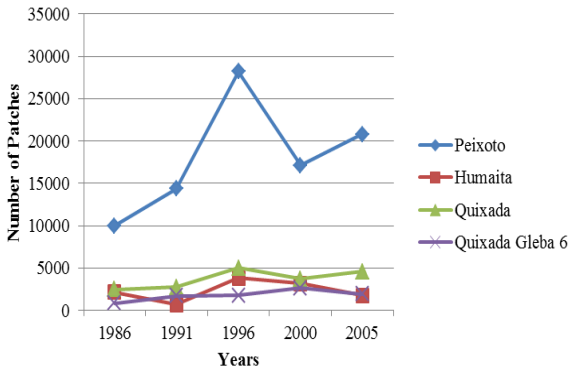

Figure 4-27. Number of patches in forest in non-forest in four PADs in eastern Acre - Brazil, 1986-2005.

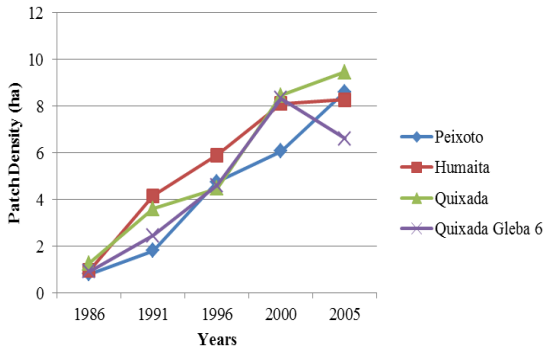

Figure 4-28. Patch density in forest in four PADs in eastern Acre - Brazil, 1986-2005.

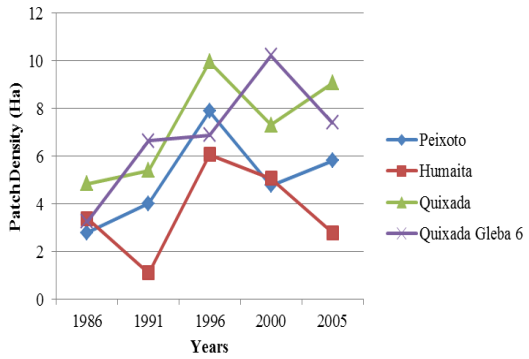

Figure 4-29. Patch density in forest and non-forest in four PADs in eastern Acre - Brazil, 1986-2005.

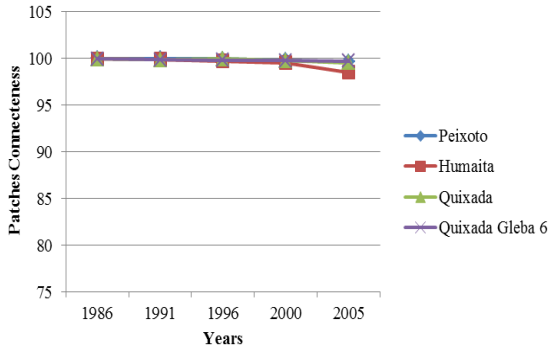

Figure 4-30. Cohesion in forest in four PADs in eastern Acre - Brazil, 1986-2005.

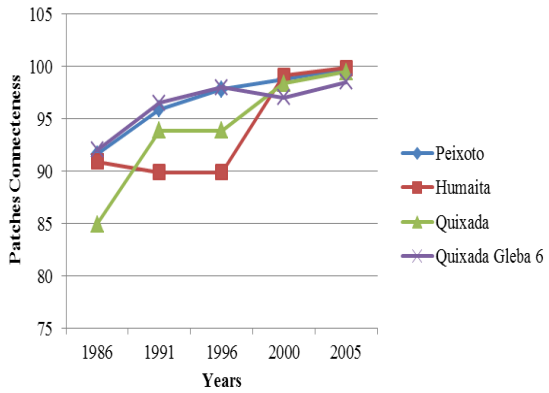

Figure 4-31. Cohesion in non-forest in four PADs in eastern Acre - Brazil, 1986-2005.

Table 4-1. Definitions of landscape metrics incorporated in the analysis of fragmentation.

Landscape Metric	Description
Largest Patch Index (LPI)	Percent of landscape comprised by largest patch
Edge Density (ED)	Edge density is the sum of all edge segments, divided by total area for each class (m/ha)
Mean Patch Area (MP-AREA)	Mean patch size (ha)
Number of Patches (NP)	Number of patches
Patch Density (PD)	Patch density, defined in terms of patches per ha (#/100ha)
Contagion (CONTAG)	Patch type aggregation, ranging from 100 (only 1 patch) to 0 (numerous patches, perfectly interspersed). Contagion drops from 1000 toward 0 as patch fragmentation proceeds and patches of a given land cover type occur less often next to other patches of the same type as they become more interspersed with patches of other land cover types.
Shannon Diversity Index (SHDI)	Represents the amount of information per patch. it is used as a relative index for comparing different landscapes or the same landscape at different times

Note: A complete description of class metric chosen in this study as well as other landscape metrics is provided at http://www.umass.edu/landeco/research/fragstats/fragstats.html.

Table 4-2. Definitions of class metric in the analysis of fragmentation of forest and non-forest.

Class Metric	Description
Percentage of the landscape (PLAND)	Percentage of the landscape on each corresponding class.
Largest Patch Index (LPI)	Percent of landscape comprised by largest patch of a given land cover class
Total Edge (TE)	Total edge length, in meters, in a land cover class
Edge Density (ED)	The sum of all edge segments, divided by total area for each class (m/ha)
Mean Patch Area (MP-AREA)	Mean patch size (ha) in a given land cover class
Number of Patches (NP)	Number of patches in a given land cover class
Patch density (PD)	Patch density, defined in terms of patches in a given land cover class per total area
COHESION	Physical connectedness among patches of a given land covers class. Values range from 100 (no isolated patches) to 0 (complete patch isolation from other patches of the same class). As patches become more clumped or aggregated, values rise.

Note: A complete description of class metric chosen in this study as well as other class metrics is provided at http://www.umass.edu/landeco/research/fragstats/fragstats.html.

CHAPTER 5
CONCLUDING REMARKS

Tropical deforestation is among the global environmental issues that have received great attention by the scientific community in the last decade due to the huge implications for the world's terrestrial carbon cycling and climate regulation. Quantifying the spatial and temporal dynamic of deforestation and forest fragmentation over time to monitor global carbon pools is therefore necessary. Deforestation and forest fragmentation present challenges to achieving sustainable development. Strategies to manage landscape change are now a matter of great interest to communities, scientists, policy makers, and to everyone concerned with the consequences of deforestation. This makes an understanding of deforestation and forest fragmentation crucial, not only for academic purposes but also for development policy goals.

This dissertation therefore seeks to better understand deforestation and forest fragmentation in Acre, Brazil, a state well-known for deforestation, innovative policy proposals for sustainable development, and diverse landscape patterns. Scientifically, this requires accurate estimation of forest and non-forest cover for specific geographic areas over time. I therefore combined satellite data and remote sensing, landscape ecology and measures of fragmentation patterns, and literature on infrastructure and land tenure to inform a three-part analysis of deforestation and forest fragmentation in Acre since the 1980s. The interdisciplinary study presented in this dissertation focuses on the analysis of both spatial patterns and temporal dynamics in land cover, both of which are necessary to adequately understand landscape dynamics. Taken together, the three papers contribute to a better understanding of deforestation and fragmentation patterns among various lands in a geographic region.

Significance of Findings

The first part of the analysis (Chapter 2) addresses the question of the sources of variability in deforestation estimates. In recent years, Acre became the focus of a broader debate over deforestation rates in Brazil under a shifting policy regime seeking to balance economic development with environmental conservation. The Government of Acre in particular promulgated a series of innovative policy proposals seeking sustainable development via forest management, but came in for national criticism by magazines reporting high estimates of deforestation rates in Acre. I therefore compared three independent sources of deforestation estimates for Acre: official estimates from INPE, and other estimates from the NGO IMAZON and an NSF project at UF. The first analytical paper pursues a systematic comparative analysis of the processing protocols used by each source of deforestation estimates. I compare techniques and decisions for radiometric calibration, geometric correction, mosaicking, and classification. This comparison serves as the basis to account for the higher deforestation estimates from INPE and IMAZON than from the UF NSF project.

The comparative analysis shows that land cover classification is the key source of differences in the deforestation estimates. The definition of what constitutes forest cover is crucial. Because "forest" has ambiguities, those ambiguities unavoidably inform definitions of forest, which can vary. In particular, in forest/non-forest classifications, the classification of ambiguous forms of vegetation, notably secondary growth, can greatly affect deforestation estimates. Whereas INPE and IMAZON classify regrowth as non-forest, UF classified older regrowth as forest. This key decision greatly helps account for why the deforestation estimates from INPE and IMAZON are higher than for UF for the same places and time points.

Hence one key conclusion is that in deforestation estimation, it is crucial to make clear how "forest" is defined. Different definitions of forest carry implications for deforestation

estimation, including political interpretations. Under specific circumstances such as those found in Brazil, where deforestation is a political question that confronts economic development with environmental conservation, choices about how to define forest are not only technical questions but also political decisions. Definitions of what counts as "forest" must therefore not only be made explicit but also come with explanations as to the implications for deforestation estimates. Otherwise, any deforestation estimates, no matter how rigorously conducted, run the risk of becoming politicized and the focus of polemic rather than understanding.

This is not however to suggest that the other steps in image processing are necessarily unimportant for deforestation estimation. While radiometric calibration, geometric correction and mosaicking were less important than classification, they are still very important to ensure data accuracy. The three data sources also varied in terms of their error accuracy and other considerations, which can become important when comparing two sources of deforestation estimates that use the same definitions of forest.

The second paper (Chapter 3) takes up the question of the causation behind deforestation by focusing on the effects of paving status, distance from highway, and land tenure type on deforestation over time. Widely-used theoretical frameworks for understanding land cover change highlight infrastructure and institutions, which are also very salient issues for land cover change in Acre. Not only has Acre recently witnessed the paving of the Inter-Oceanic Highway (BR-317 in Brazil), Acre has also served as something of a land tenure policy laboratory thru the diverse array of lands along the highway corridor, ranging from directed settlements, agricultural settlements, agro-extractive settlements, agroforestry poles, and extractive reserves. Such lands have different use rules, including different deforestation limits.

The analysis compares deforestation among the different tenure units at various locations along the highway corridor which were paved at different times, and at different distances from

the highway. While the analysis produced numerous findings and some nuances, the key findings run as follows. For the timing of paving, the findings show that regardless of location along the corridor, deforestation accelerates after paving. This corresponds to theoretical arguments that paving improves accessibility and raises land values, motivating expansion of forest clearing for agricultural activities.

For distance to the highway, the findings are weaker, in that lands more distant from the Inter-Oceanic Highway did not always exhibit lower deforestation levels or increments. This runs counter to theory on accessibility and forest clearing and much prior research. One explanation is that because the Inter-Oceanic Highway passes around the capital of Acre, Rio Branco, resulting in higher deforestation levels farther from the highway in that road segment. In other road segments, the distance gradient does appear. Another explanation is that the distance buffers may be too small (running out to 15 km from the highway), and an analysis with a larger distance buffer (to 50 km, as suggested in some prior work) might show clearer deforestation gradients.

For land tenure type, deforestation levels differ substantially between lands with higher or no deforestation limits (such as agricultural settlements and agroforestry poles) and lands with more restrictive limits on deforestation (like agro-extractive settlements and extractive reserves). Further, deforestation increments over time were greater on lands with higher deforestation limits. These findings indicate that policy experiments in tenure rules do exert a large influence on subsequent land cover change. That said, the findings also indicated evidence of rule-breaking as via higher than officially allowed deforestation in some lands (e.g., agricultural settlements) and deforestation approximating limits in other lands (e.g., extractive reserves).

The analysis also considers interactions between time of paving, distance from highway, and land tenure type. The findings there do not show strong interactions, but rather that each

factor exerts separate effects on deforestation. Hence the multivariate analysis of time of paving and tenure showed that deforestation accelerated after paving, regardless of land tenure type.

The last analytical paper (Chapter 4) addresses the question of the design of local road networks and forest fragmentation. This chapter thus takes up issues concerning public policies for rural land settlement, and how road networks affect land cover via landscape mosaics in the form of forest fragmentation. I compare four directed agricultural settlements (PADs), two close to Rio Branco and two far from the capital city. In each of these two locations, there is one PAD with a fishbone road network and one with a different type of network. I draw on theoretical insights from landscape ecology about the relationship of fragment shapes and ecological integrity in forest fragments, and I use a suite of pattern metrics to measure different aspects of fragmentation.

The analysis compares the pattern metrics for entire landscapes and forests in the four PADs for several time points to consider whether and how fragmentation varies among the road network designs over time. The first key finding is that as forest cover declines over time, forest patches became smaller, more numerous, more irregular, and more isolated (less connected), and more heterogeneous. Thus in various respects, deforestation increases both landscape and forest (class) pattern fragmentation. These findings correspond to expectations from landscape ecology and suggest a worsening situation for purposes of policies such as PES programs to encourage forest conservation and regrowth by reducing fragmentation and increasing habitat connectivity.

The other key finding in this paper concerns the effects of distance from the capital city of Rio Branco, and the design of local road networks within the PADs. In terms of distance effects, there are some differences among the landscape pattern metrics. PADs closer to the city exhibit lower edge densities and lower patch densities. While this might indicate less fragmentation, this finding actually reflects the higher deforestation levels closer to the city,

resulting in an incipient landscape saturation of deforested areas and thus less landscape heterogeneity. The findings for the class (forest) pattern metrics confirm this interpretation. There are several indications that location effects are important for the class pattern metrics. In the PADs closer to the city, there is less forest, largest forest patches are smaller, forest and non-forest edge densities are lower, mean non-forest patch areas are larger, and non-forest patch densities are lower. These findings indicate greater forest fragmentation in PADs closer to the city, and suggest that policies to encourage forest conservation and connectivity will require greater investments in those PADs than those farther from the city with less deforestation.

In terms of road network design and forest fragmentation, the findings were relatively weak. There is limited evidence among the landscape pattern metrics that road network design in PADs matters for forest fragmentation. When differences appear, they often disappear at the subsequent time point, which suggests that differences may be temporary and thus it is crucial to observe fragmentation over time. Class metrics provide some evidence that road network design affects class fragmentation. In fishbone PADs, non-forest largest patches are larger. However, the evidence is limited, which is surprising given the rather different local road networks among the PADs.

Contrary to expectations from landscape ecology, road networks do not greatly determine fragmentation patterns. This finding also bears important policy implications. Whereas the Government of Acre is developing a carbon PES program, and PES may be more useful for restoring habitat connectivity in less fragmented landscapes, these findings suggest that even with varying deforestation levels among the PADs, fragmentation does not vary greatly by road network design. This is a positive finding in that road network designs in PADs, often defined years or even decades ago without regard for the possible environmental ramifications, do not necessarily hinder the viability of implementation PES programs among designs. PES programs

should therefore be applied regardless of road network design, since the fragmentation patterns observed look no worse in PADs with fishbone designs than other designs.

Overall, the findings from Chapter 4 indicate that distance from the capital affects fragmentation more than road network design. That said, it bears repeating that fragmentation varies considerably from one time point to the next, which implies that differences observed – and the lack thereof – may be only temporary. When combined with the rising deforestation in the PADs over time, it is still possible for differentiation in fragmentation to emerge. Having said that, deforestation in the PADs has surpassed 50%, and among those closer to the city, it is approaching 70%. This could also imply future homogenization of PAD landscapes if deforestation continues, which in turn might imply reductions in fragmentation with landscape saturation of deforestation, including a reduction in possible differences in fragmentation. Unfortunately, this scenario would also imply reduced opportunities for forest conservation and regrowth as via PES programs.

Research Considerations and Future Work

Deforestation and forest fragmentation represents major challenges to achieving sustainable development and PES implementation. Many developing countries focused on information gathering as part of REDD and PES are increasingly recognizing the need for robust information not only on forest carbon and forest management activity, but also on drivers and associated trends in land use change. Assessing current and future drivers of deforestation and fragmentation is therefore very important for the design and implementation of PES strategies.

The State of Acre has emerged as a global leader in the design and implementation of innovative policies to reconcile development and conservation, as via PES programs. Especially notable has been the creation of Acre's state Climate Change Institute . A key input to PES programs in Acre as elsewhere is accurate information about the spatial-temporal distribution of

deforestation, which is necessary to estimate carbon emissions. Hence Acre's Carbon PES program will require regular updates on deforestation and fragmentation in the state.

To that end, it will be crucial for the Government of Acre to establish clear protocols for its land cover data sources, satellite image processing, and its definition of what counts as "forest." The Government of Acre should consider the merits of alternative definitions of "forest" when using any deforestation data source as well as evaluate if the definition used by the data source is consistent with their strategies to implement PES programs. A challenge is to set clear parameters concerning how secondary vegetation is treated in deforestation estimation, as regrowth accumulates carbon but also represents past deforestation.

The findings confirm that infrastructure and land tenure are important for understanding deforestation and fragmentation. While the findings for land tenure are strong, the dynamics in land cover among all tenure units observed are worrisome. The acceleration in forest loss after 2000 occurred in all types of lands, regardless of their deforestation limits. Given the completion of highway paving, I expect that deforestation will continue beyond 2005 and that differences in deforestation estimates across tenure type will remain. In other words, I do not think that deforestation will necessarily stop as legal limits are reached. The key policy question then is whether violations of deforestation limits will in turn reduce deforestation, or require governmental action. There has already been some action, such as the expulsion of residents from the Chico Mendes Extractive Reserve in 2008.

In this context, it will be important for future research to investigate if the acceleration in deforestation in Acre continues beyond 2005. This is especially important in the land tenure areas already exceeding deforestation limits, such as the PAs, but also those approaching their limits, such as the PAEs and the CMER.

In turn, new deforestation data will be very important information for the Climate Change Institute in Acre. One key issue in the implementation of the Carbon PES program is to identify areas where conservation incentives can have the largest impacts. One way to view that is to target areas that still have considerable forest, to ensure conservation; but another strategy is to target areas that have shown the largest acceleration in recent deforestation, on the expectation that those same areas are most vulnerable to larger future forest losses. More generally, the Government of Acre should also prioritize improving governance, transparency, capacity and enforcement, providing secure tenure and combatting illegal activities as foundational activities that will enable greater success in the PES implementation.

Eastern Acre also is a useful study region for an evaluation of forest fragmentation. The multi-temporal analysis shows that the four PADs eastern Acre experienced considerable deforestation and forest fragmentation from 1986 to 2005. Whereas the PADs were largely forested when created in the 1980s, they had lost half or more of their forest cover by 2005. In addition, landscape and class pattern metrics indicate considerable forest fragmentation in the PADs over the same period. Fragmentation multi-temporal analysis is potentially important for PES, and its trajectories need to be observed not just at one point in time.

Based on the analysis, it is possible to indicate that future research should emphasize fragmentation analysis since there is not much information about fragmentation in Acre. Fragmentation metrics constitute very important information that can bear important implications for targeting PES programs at specific types of lands with particular fragmentation patterns. The Government of Acre could target PADs as lands with high prospective deforestation, given recent past experience. If differences in fragmentation are limited among PADs with different road networks, then Carbon PES programs should broadly target PADs rather than only PADs with specific road network designs. To the extent that carbon PES programs should target some

types of PADs over others, they should target PADs farther from Rio Branco rather than PADs with specific types of road networks. But if the goal of carbon PES programs is to secure forest fragments in landscapes with greater patch size and connectivity, there is not a strong basis for targeting, and to the extent that there is targeting, it should focus on PADs farther from Rio Branco rather than PADs with specific types of road networks.

APPENDIX

MULT-TEMPORAL AND MULTIVARIAVE ANALYSIS OF DEFORESTATION

Table A-1. Deforestation thru time by highway paving status, selected lands along the Inter-Oceanic highway in Acre – Brazil, 1986-2005.

Lands by Time of Change in Completed Paving Status	Percent Deforested by Year (%)				
	1986	1991	1996	2000	2005
Paving by 1984 (Rio Branco-Quinari)	5.10	14.46	33.08	40.40	60.87
Paving by 1996 (Quinari-Capixaba)	4.48	14.11	11.04	13.16	34.87
Paving by 1997 (Capixaba-Xapuri)	1.53	3.44	5.41	7.63	18.52
Paving by 1999 (Xapuri-Epitaciolandia)	1.73	2.93	4.07	7.95	14.57
Paving by 2002 (Brasiléia-AssisBrasil)	5.23	9.48	10.75	17.77	23.86
TOTAL	4.21	9.04	12.64	18.30	28.44

Table A-2. Deforestation thru time by distance from highway, selected lands along the Inter-Oceanic highway in Acre – Brazil, 1986-2005.

Lands by Distance from Highway (km)	Percent Deforested by Year (%)				
	1986	1991	1996	2000	2005
0-5	5.21	15.44	18.46	25.74	37.77
5-10	2.16	6.33	9.73	15.38	25.66
10-15	5.12	6.16	10.39	14.73	23.23
Land area at all buffer area	4.21	9.04	12.64	18.30	28.44

Table A-3. Deforestation thru time by paving status and distance from highway, selected lands along the Inter-Oceanic highway in Acre – Brazil, 1986-2005.

Lands by Paving Status and Distance from Highway	Percent Deforested by Year (%)				
	1986	1991	1996	2000	2005
Paving by 1984 (Rio Branco-Quinari)					
5	3.90	11.99	24.04	32.17	54.09
10	4.68	12.98	31.13	35.82	56.18
15	6.53	20.75	42.63	51.72	71.04
Total	5.10	15.46	33.08	40.40	60.87
Paving by 1996 (Quinari-Capixaba)					
5	24.45	54.20	13.67	11.42	40.41
10	1.17	14.42	6.31	7.95	33.37
15	2.26	6.84	12.46	15.74	34.54
Total	4.48	14.11	11.04	13.16	34.91
Paving by 1997 (Capixaba-Xapuri)					
5	2.31	5.18	10.38	15.75	37.98
10	1.37	2.32	4.03	6.10	17.88
15	1.22	3.42	3.68	4.17	7.57
Total	1.53	3.44	5.41	7.63	18.52
Paving by 1999 (Xapuri-Epitaciolandia)					
5	1.54	4.08	7.24	9.10	21.75
10	3.34	4.78	6.12	11.58	21.75
15	0.72	1.61	2.35	5.48	9.05
Total	1.73	2.94	4.07	7.95	14.57
Paving by 2002 (Brasiléia-AssisBrasil)					
5	5.46	17.03	19.21	27.20	34.55
10	1.39	5.06	5.39	13.67	18.48
15	8.69	3.67	4.05	9.01	14.73
Total	5.23	9.47	10.75	17.77	23.86

Table A-4. Deforestation thru time by land type, selected lands along the Inter-Oceanic highway in Acre – Brazil, 1986-2005.

Lands by Land Tenure Type	Percent Deforested by Year (%)				
	1986	1991	1996	2000	2005
Agricultural Settlement Project (PA)	2.85	12.42	15.55	21.97	45.46
Directed Sttlement Project (PAD)	8.84	16.50	24.57	34.59	47.01
Agro-extractive Settlement Project (PAE)	1.39	2.52	2.83	5.92	9.38
Agroforestry Pole (PE)	17.74	29.63	22.65	17.50	40.13
Extractive Reserve (RESEX)	0.73	2.09	2.98	4.65	8.94
Total	4.21	9.04	12.64	18.30	28.44

Table A-5. Deforestation thru time by land tenure type and paving status, selected lands along the Inter-Oceanic highway in Acre – Brazil, 1986-2005.

Lands by Tenure Type and Paving Status	Percent Deforested by Year (%)				
	1986	1991	1996	2000	2005
Agricultural Settlement Project (PA)					
Paving by 1984 (Rio Branco-Quinari)	2.04	17.88	35.60	40.08	69.89
Paving by 1996 (Quinari-Capixaba	4.02	14.02	5.46	6.67	28.61
Paving by 1997 (Capixaba-Xapuri)	1.56	4.57	10.36	18.09	46.73
Paving by 1999 (Xapuri-Epitaciolandia)	-	-	-	-	-
Paving by 2002 (Brasiléia-Assis Brasil)	10.12	18.43	18.48	33.60	46.00
Total	2.85	12.42	15.55	21.97	45.46
Directed Settlement Project (PAD)					
Paving by 1984 (Rio Branco-Quinari)	6.18	14.61	32.20	40.52	57.69
Paving by 1996 (Quinari-Capixaba	4.62	12.24	32.27	36.62	60.03
Paving by 1997 (Capixaba-Xapuri)	10.38	17.72	20.19	31.41	40.74
Paving by 1999 (Xapuri-Epitaciolandia)	-	-	-	-	-
Paving by 2002 (Brasiléia-Assis Brasil)	-	-	-	-	-
Total	8.84	16.50	24.57	34.59	47.01
Agro-extractive Settlement Project (PAE)					
Paving by 1984 (Rio Branco-Quinari)	-	-	-	-	-
Paving by 1996 (Quinari-Capixaba	-	-	-	-	-
Paving by 1997 (Capixaba-Xapuri)	1.71	2.69	3.55	5.17	12.19
Paving by 1999 (Xapuri-Epitaciolandia)	1.79	2.27	2.77	6.96	9.60
Paving by 2002 (Brasiléia-Assis Brasil)	0.42	1.34	1.45	2.66	3.90
Total	1.39	2.52	2.83	5.92	9.38
Agroforestry Pole (PE)					
Paving by 1984 (Rio Branco-Quinari)	-	-	-	-	-
Paving by 1996 (Quinari-Capixaba	4.62	12.24	32.27	36.62	60.03
Paving by 1997 (Capixaba-Xapuri)	5.55	14.07	10.84	10.54	33.69
Paving by 1999 (Xapuri-Epitaciolandia)	17.74	29.45	22.81	13.84	41.05
Paving by 2002 (Brasiléia-Assis Brasil)	-	-	-	-	-
Total	17.74	29.63	22.65	17.50	40.13
Extractive Reserve (RESEX)					
Paving by 1984 (Rio Branco-Quinari)	-	-	-	-	-
Paving by 1996 (Quinari-Capixaba	-	-	-	-	-
Paving by 1997 (Capixaba-Xapuri)	1.21	3.54	4.85	4.74	10.12
Paving by 1999 (Xapuri-Epitaciolandia)	0.93	2.37	4.49	8.67	18.29
Paving by 2002 (Brasiléia-Assis Brasil)	0.42	1.34	1.45	2.66	3.90
Total	0.73	2.09	2.98	4.65	8.94

Table A-6. Deforestation thru time by land tenure type and distance from highway, selected lands along the Inter-Oceanic highway in Acre – Brazil, 1986-2005.

Lands by Tenure Type and Distance from Highway	Percent Deforested by Year (%)				
	1986	1991	1996	2000	2005
Agricultural Settlement Project (PA)					
5	5.75	18.07	18.44	28.00	54.12
10	1.63	9.67	11.68	16.85	41.62
15	1.33	9.79	16.46	21.13	41.06
Total	2.85	12.42	15.55	21.97	45.46
Directed Settlement Project (PAD)					
5	6.72	19.74	24.01	32.77	42.86
10	1.62	11.67	21.77	32.78	45.55
15	17.52	15.07	28.86	40.41	57.33
Total	8.84	16.50	24.57	34.59	47.01
Agro-extractive Settlement Project (PAE)					
5	1.02	3.55	5.45	8.40	13.12
10	1.38	2.12	1.75	5.08	8.12
15	1.64	2.26	2.21	5.14	8.21
Total	1.39	2.52	2.83	5.92	9.38
Agroforestry Pole (PE)	20.92	33.66	32.28	27.81	46.27
5	15.51	26.81	15.96	10.26	35.84
10	N/A	N/A	N/A	N/A	N/A
15	17.74	29.63	22.65	17.50	40.13
Total					
Extractive Reserve (RESEX)					
5	0.97	2.22	5.21	6.32	21.33
10	0.93	2.29	3.74	6.80	12.90
15	0.60	1.97	2.36	3.34	5.64
Total	0.73	2.09	2.98	4.65	8.94

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BIOGRAPHICAL SKETCH

Karla da Silva Rocha was born in 1970, in Rio Branco, Acre, Brazil. In 1988, she passed a required exam to study agronomy at the Federal University of Acre (UFAC). In April of 1993, she received her bachelor's degree in agronomy engineering. After graduation, she was invited to work as a research and extension agent with a non-governmental organization called PESACRE. During this time, she acquired a strong interest in rural development, environmental conservation and geo-technologies. In 1995, she went to the University of Florida for specialization training in Geographic Information Systems, which she used in her subsequent work to manage environmental resources. Two years later, in 1997, Karla received a Fulbright-LASPAU scholarship to work on her master's program at the University of Florida. In 2000, she completed her master's degree with a concentration in geography. Her research topic was "Application of Remote Sensing and Geographic Information System for Land-Cover and Land-Use Mapping in Pedro Peixoto Settlement in the State of Acre, Brazil." After graduation, she returned to her home country and was hired as Environment Specialist at the Environmental Institute of Acre (IMAC). Soon after, she became a professor at the Federal University of Acre where she teaches remote sensing, GIS, cartography, photo interpretation, and photogrammetry. As a professor, she received funding from the federal government through the Ministry of Education to build a GIS and Remote Sensing Laboratory in the Geography Department, which results in training of roughly 100 students every year. In 2006, Karla moved with her family to Gainesville, Florida, where she joined the School of Natural Resources and Environment (SNRE) to pursue her Ph.D. in Interdisciplinary Studies with a Concentration in Tropical Conservation and Development and Geographic Information Systems. At her dissertation defense, she presented an interdisciplinary method and accompanying theory from landscape ecology, remote sensing and Geographic Information System (GIS) to characterize spatial and temporal changes in land cover. Since

February 2010, Karla and her husband Abib have been working on their most intensive project – parenthood of two beloved girls Karol and Alana.

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