QUANTITATIVE ANALYSIS OF DEFORESTATION PATTERN DYNAMICS: DEVELOPING FOREST RESILIENCE METRICS IN AN AMAZON FRONTIER

By

JING SUN

A DISSERTATION PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
OF THE REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY

UNIVERSITY OF FLORIDA

2013

To my parents

ACKNOWLEDGMENTS

Many people helped sustain, enrich and strengthen this dissertation. Without them this document would not exist. Jane Southworth provided clear, helpful critiques on many manuscripts and greatly assisted my development as a scientist by continually confronting me with new ideas and introducing me to exciting scientists. Beyond her practical wisdom and incredible academic achievements, I came away enriched and refreshed from her humility, kindness and boundless curiosity.

My dissertation committee along with a number of other professors provided me with a wide variety of assistance. Stephen Perz introduced me to the NSF deforestation project in the Amazon, which greatly enriched my thinking on human dimensions of resilience and disturbance. Stephen Perz is an excellent sociologist, and I am honored to have had the opportunity to work so closely with him. Timothy Fik gave me a different perspective of statistics and encouraged me to think more critically. Peter Waylen always gave insightful comments and suggestions. I enjoyed working with them all.

The collegial atmosphere of the Department of Geography encouraged discussions that developed this research as well as providing the friendship necessary to support my work, and for this I would like to thank my fellow graduate students: Zhuojie Huang, Yang Yang, Forrest Stevens, Erin Bunting, Jessica Steel, Tim Fullman Huiping Tsai, Likai Zhu, Ying Wang, Qiuyin Qi, Yibin Xia, Mario Mighty, and Caroline Staub. Also, I would like to thank my friends in Gainesville, whose friendship and support helped me research and then write this dissertation.

Finally, I would like to acknowledge that this work was supported by an NSF Human and Social Dynamics Program (FY2005) project, entitled “Agents of Change: Infrastructure Change, Human Agency, and Resilience in Social-Ecological Systems”

#0527511 and also by NSF Coupled Natural Human Systems # 1114924, “Global sensitivity and uncertainty analysis in the evaluation of social-ecological resilience: Theoretical debates over infrastructure impacts on livelihoods and forest change”.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

GIS	Geographic Information System
ISO	Iterative self-organizing

Abstract of Dissertation Presented to the Graduate School
of the University of Florida in Partial Fulfillment of the
Requirements for the Degree of Doctor of Philosophy

QUANTITATIVE ANALYSIS OF DEFORESTATION PATTERN DYNAMICS:
DEVELOPING FOREST RESILIENCE METRICS IN AN AMAZON FRONTIER

By

Jing Sun

August 2013

Chair: Jane Southworth
Major: Geography

Deforestation, the process of clearing the Earth's forests, has occurred globally on a massive scale and negatively impacts the resilience of forest landscapes. Aimed to support the understanding of deforestation dynamics, the dissertation first reviews resilience theory, and then demonstrates possible solutions to infer the resilience of deforested landscapes with particular attention to the forest spatial pattern changes via an example in the Amazonian rainforest. The solutions were based on the application of three different methods, and integrating these with remote sensing (RS) and geographic information system (GIS) techniques. Data inputs were fine-grained forest/non-forest classifications of the years 1986, 1991, 1996, 2000, 2005 and 2010 interpreted from Landsat satellite images. The three different methods presented in this dissertation are: first, the combination of a moving window approach and an iterative self-organizing (ISO) unsupervised classification used to describe the ‘cross-boundaries’ influences of deforestation. These patterns are revealed by forest cover changes, which are initiated from developed areas and then extend into forest areas across multiple spatial scales. Second, the local changes of forest spatial patterns were illustrated, where the dynamics of forest cover were obtained by applying the morphological spatial pattern

analysis (MSPA). Two additional MSPA tools, ‘hole’ and ‘outer background’, were created to enhance the appreciation of the spatial patterns of developed areas. Third, fractal geometry was introduced to further prompt the understanding of the spatial patterns of developed areas. A fixed-grid scan strategy was adopted to pixelize the entire landscape and to cartographically represent the spatial patterns of developed areas by fractal dimension. Moreover, a configuration framework was proposed to partition the level of deforestation characterized by fractal dimensions and as such can potentially be used to better understand and manage the evolution of forest clearings. These three methods each have their own set of advantages and limitations, and yet work in a complementarily manner and cast particular insight on spatial pattern change during the process of deforestation. Their useful value-added contributions towards resilience theory is discussed at the end, as the ontogenetic stage of resilience theory is moving from the ‘consolidating’ stage to the ‘empirical interactive’ stage.

CHAPTER 1
INTRODUCTION

Geographers study the spatial and temporal distribution of phenomena, processes, and features as well as the interaction of humans and their environment. Because space and place affect a variety of topics, such as economics, health, climate, plants and animals, it is necessary to introduce a framework to integrate multiple topics in an interdisciplinary system. The recently discussed resilience theory might be able to tackle and build critical thinking towards this question, studies on social-ecological resilience in geography (Agder, 2000, 2003) as well as on more or less successful responses to disasters, such as Hurricane Katrina in 2005, have greatly raised interest in the use of resilience theory.

Operating within the paradigm of resilience theory, this dissertation investigates the resilience of deforested landscapes from a combination of theoretical, technique and empirical perspectives. The approach followed is to build a set of models to understand the resilience of forest landscapes to deforestation activities, which has been completed in this dissertation, in order to then provide the information to conservation programs and then guide real-world forest management. The dissertation is structured into seven chapters. This brief introduction to the organization of the dissertation is Chapter 1 and the remaining chapters are outlined below:

In Chapter 2, resilience theory and its development are briefly introduced. The necessity of applying resilience theory in deforestation studies is discussed and then proposed as the research paradigm of this dissertation.

Chapter 3 introduces the reader to the data used throughout the upcoming analyses and describes the procedure followed for the satellite images classification

and processing. An object-oriented classification method, a decision tree approach, was applied, which largely improves the classification accuracy compared with more traditional techniques. The products are multiple forest/non-forest maps of the years 1986, 1991, 1996, 2000, 2005 and 2010, which are utilized as the only data source for Chapters 4, 5 and 6 which are themselves outlined and discussed below.

The pattern and process of deforestation are not restricted to cleared areas but rather they exhibit strong influences beyond the clearings physical boundaries. Such that, Chapter 4 presents a moving window technique with an ISO supervised clustering algorithm, this combination is designed to analyze landscape information across multiple spatial scales by varying window sizes, which help think through the structure, function and dynamics of deforested landscapes from local scales to regional scales.

Deforestation affects the spatial layout of all components of the forest system, and spatial patterns have been shown to influence many processes that are ecologically important. Therefore, quantitative methods are required to detect spatial patterns of deforestation, identify significant changes through time, and relate these patterns to important ecological functions. So that, Chapter 5 presents an image processing tool, MSPA, as used here to describe the process of landscape fragmentation as deforestation proceeds. It is described in detail and compared with the frequently used FRAGSTATS metrics. Importantly, two new MSPA tools, ‘hole’ and ‘outer background’, are created and proposed to further classify the spatial patterns of developed areas. The incorporation of these two new tools improves the MSPA algorithms, expands their potential applications, and provides a new perspective to understand the spatial patterns of developed areas as well as the entire landscape.

Chapter 6 introduces the concept of fractal geometry as an application for studying the spatial patterns of developed areas. Specifically, the heterogeneous landscape is pixelized by a fixed-grid scans approach and then the value of each pixel is characterized by the fractal dimension calculated by the box-counting method. The results indicate that deforestation has expanded across the landscape but with different patterns and rates. Such differences are summarized within a proposed framework which indicates the level of deforestation or development, and as such, can potentially be used to apprehend and regulate deforestation over time.

Chapter 7 summarizes and concludes the dissertation. Critically, a new development in the philosophy of science inspired by resilience theory is discussed, utilizing the research presented.

CHAPTER 2
RESILIENCE AND DEFORESTATION IN THE AMAZON

What Is Resilience?

Development of ecological resilience theory began in the 1960s with attempts to mathematically model ecosystem dynamics (Gunderson et al., 2010). Early models focused on the stability of systems, such as Lewontin (1969) and May (1977), where they assumed ecological systems were stable from both theoretical and practical perspectives. Usually, their research emphasized a global or stable equilibrium, such as carrying capacity in ecosystem, and usually led to policies of maximum.

Holling (1973) introduced the word resilience to describe three aspects of changes that occur in an ecosystem over time. The first was to describe the "persistence of relationships within a system" and the "ability of systems to absorb changes of state variables, driving variables and parameters, and still persist". The second concept recognized the occurrence of alternative and multiple states as opposed to the assumption of a single equilibrium and global stability; hence, resilience was the amount of disturbance a system can absorb before it shifted into alternative configurations. The third insight was "the surprising and discontinuous nature of change", such as the collapse of fish stocks or the sudden outbreak of spruce budworms in forests. These insights altered the way in which theorist's perceived ecological systems and how practitioners have attempted to manage these systems.

The Promise of Ecological Resilience as a Theoretical Framework for Land Use and Land Cover Change

Resilience has two faces, engineering and ecological (Holling, 1996). Engineering resilience is defined as the rate or speed of recovery of a system following a shock (Pimm, 1984; O'Neill et al., 1986; Tilman and Downing, 1994). Ecological

resilience, on the other hand, assumes multiple states (or “regimes”) and is defined as the magnitude of a disturbance that triggers a shift between alternative states (Holling, 1996). In this sense, a regime shift occurs when the controlling variables in a system (including feedbacks) result in a qualitatively different set of structures and dynamics of these systems (Walker et al., 2004).

It has been four decades since the term resilience was introduced, and various documentations still maintain momentum. Scientists, working in disturbance-driven ecosystems, found that resilience theory was the only theory that helped explain the complex changes that they were studying (Walker et al., 2004). Scheffer et al. (2001) described two alternative states, clear water with rooted aquatic vegetation and turbid water with phytoplankton, in shallow lake systems. Coral reef systems may shift from a coral-dominated state to a macroalgae-dominated state (Hughes, 1994). Terrestrial systems have traditionally been studied in much more detail than most freshwater or marine systems; and as such, there is a large collection of compelling examples of critical transitions between alternative states. Walker (1981) studied semiarid rangelands and found dramatic shifts between grass-dominated and shrub-dominated systems. Those shifts were mediated by interactions among herbivores, fires, and drought cycles. Gragson (1998) argued that disturbances derived from land use and land cover changes are related to the extent of the transformation, and he recognized two types: modification, a change within a given cover type (e.g., from closed to open forest), and conversion, that is, a change from one cover type to another (e.g., forest to crop), which involves a more profound transformation and, therefore, an associated change in resilience. Foster et al. (1998) suggested that forest cover loss has a

substantial effect not only on the structure and functioning of ecosystems, but also on resilience and management. Shifts in forest cover, associated with management of fire regimes, in the well-drained soils of the southeastern United States reflect alternate states.

Among these examples, not all threshold-crossings are sudden. Evidently, most terrestrial systems “flipped” from one attractor to another with very gradual changes. This does not indicate these gradual changes are not important. Conversely, in many cases, regime shifts may be largely irreversible (Carpenter et al., 2009; Scheffer, 2009). For example, Wilson and Agnew (1992) demonstrated that the forests were established under a wetter rainfall regime thousands of years previously. Necessary moisture is supplied through condensation of water from clouds intercepted by the canopy. If the trees are cut, this water input stops and the resulting conditions can be too dry for recovery of the forest. Similarly, extinction of mega-fauna at the end of the Pleistocene in Siberia, possibly through improvement in hunting technology, may have triggered an irreversible shift from steppe grassland to tundra. The resulting increase in mosses led to cooler soils, less decomposition, and greater carbon sequestration in peat (Zimov et al., 1995).

Deforestation in large forest areas, like the Amazonian rainforest, coupled with the effects of regional climate change, are pushing the rainforest towards a “tipping point” (Scheffer, 2009) where it would irreversibly start to die (probably being turned into savanna or desert), with catastrophic consequences for the world’s climate (Scheffer et al., 2001, 2007). In order to prevent the (undetectable, unforeseeable) irreversible shift

in this forest system, it is necessary to build models to appreciate deforestation dynamics and most importantly, system resilience.

Understanding Deforestation: the Importance of Modeling

Deforestation is the process of clearing the Earth's forests and has occurred globally on a massive scale which can negatively impact the resilience of forest systems, resulting in numerous repercussions. Understanding deforestation is important to regional biodiversity and global climate. The large-scale estimate of deforestation has been feasible since the analysis of satellite imagery which began in 1972, and there is a large collection of compelling examples of deforestation studies (Southworth and Tucker, 2001; Brook et al., 2003; Nagendra, 2004). Deforestation, most often, goes with economic development and is continuing today into the remaining areas of undisturbed forest globally. Time-series satellite data confirms that due to human land use, deforestation is already underway in the Amazon and will probably continue well into the future.

Road building is a key determinant of deforestation in the Amazon (Perz et al., 2008). The roads built in the Amazon area can be classified as two types: ‘official’ roads and ‘unofficial’ roads (Perz et al., 2007a, b, 2008). Official roads were built by the state as part of the development policy to meet geopolitical and economic objectives. Paving of official roads raises land values, which provides the incentive to exploit natural resources farther out from official road corridors and consequently motivates unofficial road building. Unofficial roads were built by local interest groups specifically to gain access to land, timber and other natural resources. Unlike official roads which were published on maps or regulation layouts, the building of unofficial roads does not have official documentation or planning. Unofficial roads keep growing rapidly, and are not

sustainable (Perz et al., 2005, 2007b). Unofficial roads form dense networks and these fragment forests into small, irregular shapes, and importantly diminish the resilience of forest systems (Peterson, 2002).

Deforestation processes, especially when these processes occur inside the forested areas, can disconnect conterminous forest areas which largely change the spatial properties of forest systems, such as forest connectivity. The consequences affect the forest health and forest succession. Forests are habitats to animals and wildlife and therefore the reduced forest connectivity would limit the food sources and the behaviors of these species, and probably lead to extinctions in a few decades. The highly isolated forests were not able to maintain material and energy flow, which would inevitably threaten the function of ecosystems and human society. However, deforestation keeps expanding with population growth and economic development in Amazonian nations, hence, we need to analyze this increasingly fragmented forest landscape, otherwise the whole forest landscape with its ecosystem may be going to a catastrophic shift which will inevitably cause economic and ecological losses.

When viewing the spatial pattern of a certain landscape, we look at its composition and spatial configuration: the elements present and how these elements are arranged (Li and Reynolds, 1994; Turner et al., 2001). In deforestation landscapes, we may observe complex patterns of forest and non-forest surfaces (like fish bone or chess board), we may observe gaps in an otherwise continuous forest, and we may detect large contiguous areas of deforestation, reforestation and “mature” forest. Deforestation patterns affect the spatial patterning of all components of the system, and have altered the resilience of historical landscapes (Gunderson and Pritchard, 2002).

Understanding forest resilience is critical for managing deforestation, and for restoration and preservation of historical landscapes. But, it is difficult to directly measure resilience (Carpenter et al., 2005), so that a set of models is proposed in this dissertation to study deforested landscapes and demonstrate it as an indirect measure to indicate forest resilience (to deforested disturbances). In order to accomplish this, the dissertation provides three measurements to evaluate the spatial patterns of forest landscapes from different perspectives. The modeling would help to gain an understanding of how deforestation affects the composition and configuration of forest landscapes, as well the function and services of the forest, and as an indirect measure of system resilience. It is worth noting that 'knowledge transfer' is the dominant objective threading through the whole research, as the results are expected to provide useful knowledge in forest management and conservation programs.

Deforestation in the 'MAP' Amazonian Rainforest

Over a 25-year period, as a function of distance to major roads, forest conversion occurred in the Amazon region of South America, the largest remaining tract of contiguous tropical forest in the world. The study area of this dissertation is restricted to the heart of southwestern Amazon, located near the shared borders of the Peruvian state Madre de Dios, Acre in Brazil, and the Bolivian department of Pando (termed the 'MAP' region, approximately 300,000 km^2, see Figure 2-1). This tri-national frontier was historically covered by humid tropical forest and remains roughly 90% forested. It is home to the world's highest levels of terrestrial biodiversity and is considered a global biodiversity 'hotpot' (Myers et al., 2000; Killeen and Solórzano, 2008). On the other hand, MAP is a relatively isolated area in three countries and is featured by a composite of rich natural resources and fairly poor economic development. Such differences in the

MAP frontier and elsewhere in South America prompted presidents of several countries to develop the Initiative for the Integration of Regional Infrastructure in South America (IIRSA). IIRSA prioritizes cross-border infrastructure projects such as road paving as a means of economic integration and economic development.

The MAP frontier is among the first wave of IIRSA projects, and is currently being integrated via paving of the Inter-Oceanic highway. The paving in Acre was completed in 2002 and therefore forest clearings are more extensive and numerous in this region, while the paving in Madre de Dios began around 2005 and is only now being completed, so that the deforestation in Madre de Dios is relatively patchier than that of Acre. On the other hand, the road in Pando has not been paved yet, and so here most forest clearings occur around major towns and along the Brazilian border (Southworth et al., 2011). Thus, there is considerable spatial variability in land cover changes and the three regions exhibit a pathway of development from most developed (Acre), to developing (Madre de Dios), and currently much less developed (Pando). This is a critical historical moment at which to obtain estimates of past and present land cover change in order to see how the MAP social-ecological system responds to external disturbances brought by migration, investment and new technologies in the wake of infrastructure upgrades (Brown et al., 2002; Perz et al., 2011).

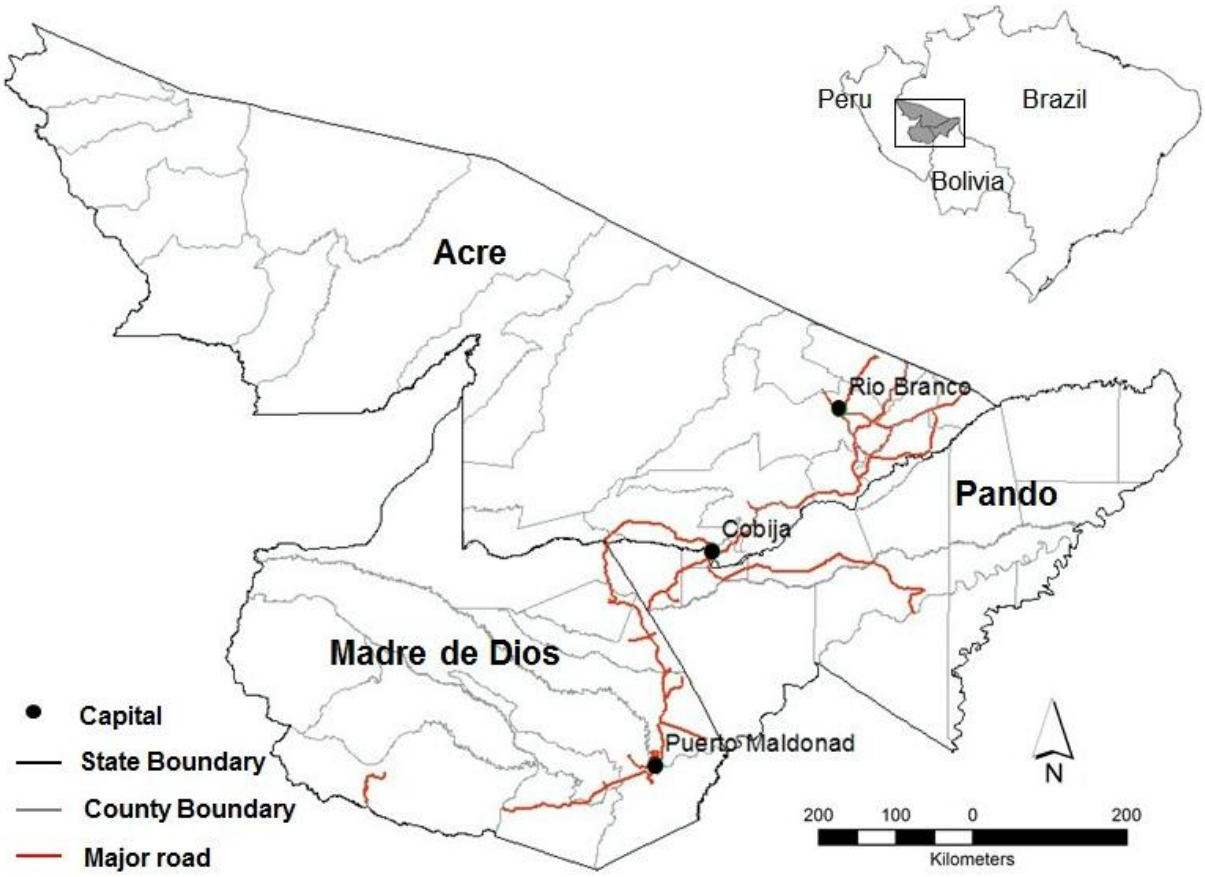

Figure 2-1. Map of the study area with major roads and state capitals superimposed. The region encompasses the tri-national frontier of the Peruvian state of Madre de Dios, the Brazilian state of Acre, and the Department of Pando, Bolivia (termed the 'MAP' region).

CHAPTER 3
DATA SOURCE: FOREST/NON-FOREST COVER MAPS

A land cover database from digital maps of the land surface, documents precisely where land cover change has occurred, which is used for thousands of applications in diverse investigations such as ecosystem status and health, spatial patterns of biodiversity, indications of climate change, and best practices in land management (Turner et al., 2001). In the MAP region, deforestation is currently expanding and therefore, it is important to make a set of land cover databases to record the detailed information of deforestation processes and then to analyze their spatial patterns. Since deforestation is a dynamic process, the cost of gathering timely and accurate information becomes prohibitive if done by field work alone. In addition, deforestation is a spatial-temporal process with categorical change (forest and non-forest) and requires studying through temporal ordering, which made satellite data become a major data source in the large-scale forest research.

Landsat Satellite Data

Single-layered binary deforestation index maps are derived from Landsat TM 4, 5 (TM is the abbreviation for thematic mapper) and ETM + 7 (ETM + is the abbreviation for enhanced thematic mapper plus) satellite images across six near-anniversary months with less than 10% cloud for detection (July or August, 1986-1991-1996-2000-2005-2010, corresponding to the dry season that permits minimal cloud cover and lower aerosol concentrations).

Landsat 4 is the fourth satellite of the Landsat program and was launched in 1982. Landsat 4 science operations ended in 1993 when it lost the ability to transmit science data, far beyond its designed life expectancy of five years. Landsat TM 5 is the

fifth satellite of the Landsat program and was launched in 1984. Landsat TM 5 is still sending back data now, and in the year 2012, it is 25 years over its 3-year mission. The satellite was said by the USGS (USGS is the abbreviation for United State Geological Survey) to be nearing the end of its life, after more than 28 years in space. Landsat 7, launched in 1999, is the latest satellite of the Landsat program. Landsat 7 is the most accurately calibrated Earth-observing satellite, i.e., its measurements are extremely accurate when compared to the same measurements made on the ground. The Landsat 7 mission went flawlessly until May 2003 when a hardware component failure left wedge-shaped spaces of missing data on either side of Landsat 7's images. The remainder of the ETM+ sensor, including the primary mirror, continues to operate, radiometrically and geometrically, at the same high-level of accuracy and precision as it did before the anomaly; therefore, image pixels are still accurately geolocated and calibrated.

Two data providers were used, the USGS's Earth Resources Observation and Science Center (EROS, http://glovis.usgs.gov) and Divisão de Geracão de Imagens (http://www.dgi.inpe.br), all scenes were freely downloaded from two websites to comprise the MAP region (path/row of the scenes: 1/67, 1/68, 2/67, 2/68, 2/69, 3/67, 3/68, 3/69) for the target years.

Image Pre-processing

For each image, we performed the atmospheric and radiometric calibration, standardization of map projection, coordinate system, and datum; and geometric correction to the Global Land Cover Facility Geocover product for 1999/2000. The image preprocessing follows the CIPEC protocol (Moran and Ostrom, 2005) and was standardized for each image, providing a robust preprocessing methodology

comparable across dates and country boundaries and cut to exclude clouds and rivers, and undertook an object-oriented classification to produce forest/non-forest maps. Due to the similarity of spectral signatures, some transitional woodlands and agro-forestry areas are difficult to separate and have been classified as forest. Agriculture, cleared areas, major road, urban fabrics, and shrub were aggregated to create a non-forest class.

Image Classification: Object-oriented Approach

Image mosaics were created for 1986, 1991, 1996, 2000, 2005 and 2010 with eight images for each year. Explicatory variables: tasseled cap brightness, greenness, and wetness indices were calculated along with a mid-infrared vegetation index and a three-by-three moving window variance for each mosaic date as input to a decision tree classification. Due to striping in the visible and thermal bands in many of the available Landsat images, only the near- and mid-infrared bands were used along with the derived products to create a forest and non-forest classification for each mosaic date (see Marsik et al., 2011 for more detail on processing of imagery). The image striping problem limited the spectral data available, and due to this problem, we created derived products, rather than use much of the raw image bands, as these derived products did not have the striping problem (e.g., TCA analysis which is a form of Principal Components Analysis (PCA) itself a technique commonly used to destripe images, and so the first three components of the TCA were striping free, and used in concert with the unaffected image bands). The forest/non-forest classification was used in this analysis due to the focus on deforestation processes as a function of road building. Differences in timing and type of agriculture made a set agricultural class inaccurate given the available imagery. As such pasture, bare fields and built cover types were grouped into

the non-forest class. The forest class included all dense vegetated covers, which would by default include secondary succession as a cover type once a dense canopy was achieved (considered to be 3–5 years within this region). A decision-tree classifier was used for the analysis to allow the incorporation of a series of rules and derived image products within the classification. Compumine Rule Discovery System (RDS) software was used, which is a data mining software which predicts the specified land cover classes, and was established here using a split-sample validation, where we used 85% of our training sample points to create the decision rules and tree classifier and the remaining 15% were used to test the tree output. The rules, once developed and tested for accuracy (each year was analyzed separately and percentage accuracy of the rules were 98–99.8% accurate, see Marsik et al., 2011 and Southworth et al., 2011 for more detailed information) were then incorporated into the ERDAS Knowledge Engineer rule-based classifier to create each year‘s land cover classifications. Classification accuracy was assessed using over 350 training samples collected during fieldwork from 2005 to 2006 and Kappa coefficient and overall percent accuracy for each class and for the overall classification, with resulting accuracies for 2005 of greater than 90%. In addition the 2000 image data were independently checked using ASTER images for the year 2000, and an overall accuracy of 96% was achieved across these products. The 2010 image data were also independently examined using Google Earth, and an overall accuracy of 95% was reached across the products. So that, these six forest/non-forest binary maps are the inputs for the sections outlined and discussed below.

Forest Cover Change in the MAP Region

With relatively uniform temporal interval and fine spatial resolution, the created time-series forest/non-forest cover maps (Figure 3-1) will be able to reflect the change

trajectories of forest covers, forest areas, fragmentation patterns (Tole, 2006) and other forest landscape properties. Specifically, from 1986 to 2010, we observed an increase in non-forest area for all three states (Acre, Madre de Dios, and Pando), at the expense of forest areas. Across all three regions, the forest is still really dominant and the landscape is quite static, such that, in 2010 the forest cover percentage is 80.0% in Acre, 97.4% in Madre de Dios, and 98.0% in Pando (Table 3-1). Correspondingly, the area percentage of the non-forest area is relatively small, with Acre at 20.0%, Madre de Dios at 2.6%, and Pando at 2.0%.

Table 3-1. Summary of forest cover changes from 1986 to 2010 in the MAP region*

Region	Cover	1986(%)	1991 (%)	1996 (%)	2000(%)	2005(%)	2010(%)
Acre	Non-forest	2.5	6.6	7.5	9.6	14.4	20.0
	Forest	97.5	93.4	92.5	90.4	85.6	80.0
Madre de Dios	Non-forest	2.2	1.7	0.9	1.8	2.0	2.6
	Forest	97.8	98.3	99.1	98.2	98.0	97.4
Pando	Non-forest	0.4	0.7	0.8	1.3	1.9	2.0
	Forest	99.6	99.3	99.2	98.7	98.1	98.0

*where 1% in Acre represents 622.79 km2, 1% in Madre de Dios represents 493.48km2, and 1% in Pando represents 542.87 km2.

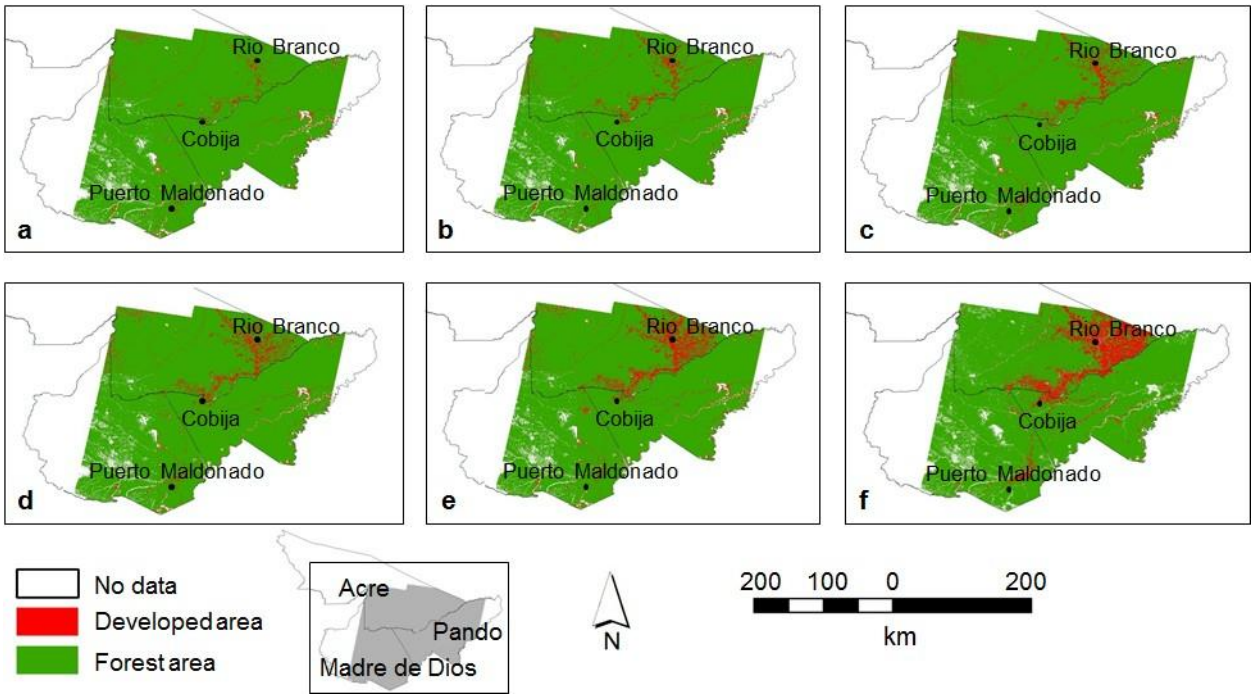

Figure 3-1. Forest cover maps showing fragmentation dynamics in the MAP region for A) 1986. B) 1991.C) 1996. D) 2000. E) 2005. F) 2010.

CHAPTER 4
BEYOND THE BOUNDARIES: MAPPING 'HALO' IMPACTS OF DEFORESTATION

Background

Land use and land cover changes play a major role in the study of global change, which have largely resulted in deforestation, biodiversity loss, global warming and increase of natural hazards (Mas et al., 2004; Dwivedi et al., 2005; Reis 2008). Significant attention has been drawn to areas of tropical forest cover, and specifically the Amazon region across the past few decades due to the alarming rates of clearing which have been occurring here (Nepstad et al., 1999, 2001; Anderson et al., 2002; Giles and Burgoyne 2008; Marsik et al., 2011; Southworth et al., 2011). Laurance et al. (2004) pointed out that infrastructure development projects are significant contributors to deforestation in the Brazilian Amazon, which have caused numerous complex problems to social-ecological systems (Cumming et al., 2012). In forest landscapes, like the Amazonian rainforest, a new and pervasive spatial pattern has been created by road paving (Riitters and Wickham, 2003) and then been enhanced by the expansion of human settlements. The road paving encourages economic developments via regional integrations, such as facilitating ties of remote areas to local and regional markets (van de Walle, 2002) and opens up previously inaccessible and intact rainforest to these more developed regions. To access the natural resources, the paving of major roads stimulates the construction of secondary and tertiary level roads, as well as 'unofficial', often temporary logging roads which continue to be used well after their initial creation and often become permanent landscape features (Perz et al., 2007a, b, 2008), and these roads form dense and hierarchical networks where the land cover changes along them exhibit some recursive structures.

For years, economic, social, and ecological research has drawn contrasting conclusions about the consequences of such new patterns created by road paving (Perz et al., 2012). This infrastructure is notorious for intensifying social conflicts over newly accessible natural resources (Schmink and Wood, 1992). For example, lack of participation raises issues of exclusion of indigenous and other minority groups, often whereby newcomers make legalistic claims to natural resources in order to discredit traditional claims (Perz et al., 2012). However, both macroeconomic and microeconomic literatures demonstrate the positive impacts of road paving on the economy, specifically empirical studies show significant poverty reduction near roads, whether via analysis of sub-national administrative units (Demurger, 2001; Fan et al., 2004), local communities (Gunasekera et al., 2008), or individual households (Gibson and Rozelle, 2003). On the other hand, there are significant ecological changes which can occur as, both road networks and developed areas could fragment a continuous forest into discrete habitats and thereby reduce landscape connectivity (Estrada and Bodin, 2008). Connectivity is a vital element of landscape structure (Taylor et al., 1993), and one to which the persistence of spatially structured species populations, and meta-populations, is strongly related (Hanski and Ovaskainen, 2003). Lack of connectivity reduces the capability of species movement (i.e., local extinctions of certain species could not be compensated by the immigrant population from the neighbor patches) and interferes with pollination, seed dispersal, wildlife migration and breeding (Estreguil and Mouton, 2009). As such, via road networks, the impacts of social conflicts, economic benefits, and ecological damages extend from a small-scale region (cleared area) to an entire social-ecological system. Such ‘halo’ impacts (Wade et al., 2009) demonstrate that the

pattern and process of deforestation are not restricted to those cleared areas but rather they exhibit strong influences beyond the clearings physical boundaries. As such, a multiple scale analysis that includes larger surrounding areas beyond just the clearings themselves becomes a key factor to better understand the social-ecological systems in deforested landscapes.

One approach to understanding such ‘halo’ impacts is based on the patch-corridor-matrix model (Forman, 1995) that simplifies the landscape structure as a network with individual patches connected by corridors (Hanski, 1999; Opdam, 2002). Thus, a landscape can be described as an area of land containing a mosaic of habitat patches, often within which a particular focus habitat patch is embedded (Dunning et al., 1992). Zaccarelli et al. (2008) discussed and applied the dualism phrase ‘content and context’ to explain the multiple scale analysis with a conservation example in Apulia, Italy. Specifically, they explained the ‘contents’ of the network by descriptions of patches and corridors within the boundaries of the conservation area, where each area is assumed to be spatially homogeneous according to the corresponding descriptions. In contrast, the ‘context’ of conservation areas refers to the nature of the surrounding landscapes that may exhibit significant impacts on interiors within the conservation boundaries (Wiens, 2002). At a landscape level, the context can be taken as the buffer zone around a target area, which exhibits either negative or positive impacts on the content, as different land uses and land covers may promote or block ecological function and services, such as population migration (Hansen and Rotella, 2002). To further illustrate the interactions between the content and context, Riitters et al. (2009) proposed a ‘forest security’ concept to interpret forest dynamics at a pixel level and

defined the forest security as the likelihood that a pixel of forest remains as forest over time, which depends on the surrounding landscape mosaic. The security level of context that surrounds the content is not uniformly distributed, in fact, the security level declines when the distance to the content increases. During the expansion process of developed areas, the interfaces between forest areas and non-forest areas, e.g. forest edges, become sensitive areas that usually have a high likelihood to be converted into non-forest. Such interface was not emphasized in the classic landscape analysis before (Cumming et al., 2012), but it has been estimated that the amount of Amazonian rainforest modified by the forest edge exceeded the area that had been cleared (Skole and Tucker, 1993). As such, content, context, interface (i.e., forest edge), and forest security level should become important considerations in any multi-scale analysis of deforested landscapes.

Landscapes do not exist in isolation. Landscapes nest in a panarchic structure (Gunderson and Holling, 2002; Zurlini et al., 2006), that is, focus landscapes embed within larger landscapes that embed within even larger landscapes, and so on. This panarchic representation makes the multi-scale analysis imperative for thinking through structure, function and dynamics of deforested landscapes (Wu et al., 2000). In answering such a requirement, moving window operations provide a useful technique to derive multi-scale information in landscape research (e.g. Riitters et al., 2000, 2002; Southworth et al., 2006; Zurlini et al., 2006; Zaccarelli et al., 2008; Wade et al., 2009), which would be able to appreciate and then link the local pattern and process of deforestation to the regional ‘halo’ impacts that occur beyond the boundaries of extant cleared areas. In this section, we will apply a moving window operation combined with

the ISO unsupervised classification to understand the ‘halo’ impacts of deforestation in the MAP region.

Moving Window Analysis across Multiple Spatial Scales

Advancements in satellite technology combined with mathematical and/or spatial statistical methods have been widely used in landscape ecology. To measure the ‘halo’ impacts on forest/non-forest binary maps from 1986 to 2010, we used a moving window algorithm to locate a set of 9 fixed-area windows around each pixel. The window sizes are 0.15, 0.39, 0.81, 1.65, 3.21, 6.45, 9.75, 12.87, and 16.11 kms on a side, where the window side lengths span two orders of magnitude (from 0.15 km to 16.11 km) to ensure, as best as possible, accurate estimates of the scale-dependent behaviour of percentage fragmentation (Wade et al., 2009). We measured the amount of fragmentation within each window by the proportion of non-forest pixels (P_n) on each binary map, and then produced a stack of these raster layers, where each raster was the specific output for a certain window size. Next, we performed the ISO clustering algorithm (also known as the migrating means technique) to run a clustering analysis according to the similarity of P_n values over all 9 window sizes, where ISO is an abbreviation for the iterative self-organizing way of performing clustering (Zurlini et al., 2006; Zaccarelli et al., 2008). The ISO Cluster algorithm is an iterative process for computing the minimum Euclidean distance when assigning each candidate cell to a cluster. The process starts with arbitrary means being assigned by the software (ArcGIS 10.0), one for each cluster (the cluster number is defined by the user). Every cell is assigned to the closest of these means (all in the multidimensional attribute space). New means are recalculated for each cluster based on the attribute distances of the cells that belong to the cluster after the first iteration. The process is repeated: each cell

is assigned to the closest mean in multidimensional attribute space, and new means are calculated for each cluster based on the membership of cells from the iteration. The number of iterations of the process is specified by the user, and this number should be large enough to ensure that, after running the specified number of iterations, the migration of cells from one cluster to another is minimal, and thus all the clusters become stable.

Corresponding to the previous discussion of content, context, forest edge, and forest security level, we defined five clusters in this research: core forest area, sub-forest area, transition area, sub-developed area, and core developed area. For the forest areas (content), the security level of the core forest area is the highest and then the security level decreases from the sub-forest area to the most insecure level, which is the transition area. Meanwhile, we can interpret cleared areas (context) using a similar gradient level: the core developed area where forest clearing is permanent, and then the sub-developed area where the forest clearing is new, and again the transition area where the forest clearing is most likely underway. As such, the 'halo' impacts of deforestation start from the core developed area, spread over the sub-developed area, via the transition area, and finally reach the sub-forest area (see Figure 4-1 for a detailed illustration of the method described). To validate this classification framework for the real world landscape we will implement this moving window operation to study deforestation dynamics in the MAP region from 1986 to 2010.

Results

'Halo' analysis

To represent the 'halo' impacts at multiple spatial scales, the results of the clustering analysis are summarized in Figure 4-2 and Table 4-1. Visually, the 'halo'

impacts of the cleared areas in the MAP region (core developed area, sub-developed area, transition area, and sub-forest area) expand rapidly from 1986 to 2010, and the developed areas increase substantially among them. In Acre, the core developed area increases from 1.7% in 1986 to 13.3% in 2010 (where 1% in Acre represents 622.79 km2), which may signal the beginning or initialization of the opening up (urbanization) of this predominantly forested landscape. Subsequently, its impacts spread over the surrounding classes, where the sub-developed area increases from 3.0% in 1986 to 6.7% in 2010. In addition, the transition area has a slight increase from 4.4% in 1986 to 6.7% in 2010, since this interface area usually has elevated temperature, reduced humidity, and increased wind speed relative to forest interiors (Cumming et al., 2012) the expansion of this area would bring many uncertain factors to the forest system. The sub-forest area decreased from 15.9% in 1986 to 14. 7% in 2010 and the core forest area changes from 75.0% in 1986 to 58. 6% in 2010, and such significant reduction of these two stable forest classes indicates the entire forest landscape in Acre may be undergoing a critical transition. In Pando, the core developed area increases from 0.2% in 1986 to 2.0% in 2010 and the sub-developed area changes from 0.5% in 1986 to 2.5% in 2010 (where 1% in Pando represents 542.87 km2), and the small percentage of these two non-forest clusters (4.5%) until 2010 characterizes the less-developed situation in Pando. However, there is a re-allocation of the forest area among the three clusters, where both the transition area (0.6% in 1986 to 3.3% in 2010) and the sub-forest area (5.5% in 1986 to 18.5% in 2010) increase significantly, while the core forest area decreases from 93.3% in 1986 to 73.7% in 2010. Though the total forest area (transition, sub-forest, core forest) is still high in 2010 (95.5%), there is a large part of

the forest area transformed from stable core forest to the unstable clusters: transition and sub-forest. On the other hand, Madre de Dios experienced a reforestation-deforestation process during the past 25 years and therefore the percentage area of the core forest starts to increase from 75.1% in 1986, reaches the peak at 87.0% in 1996, and finally decreases to 75.7% in 2010. Consequently, the other four clusters: core developed, sub-developed, transition and sub-forest have a U-shape change from 1986 to 2010. The use of these more detailed 'halo' analyses reveals significant landscape changes in forest cover across these regions (as illustrated in Table 4-1) that if studied in a more traditional or static analysis is completely obscured. Omission of material that unites components to form a complete manuscript.

Clustering across scales

For the three states in the MAP region, the profiles of the five cluster means along multiple window sizes were depicted (Figures 4-3, 4-4, 4-5). For a given location, the profile of Pn with different window sizes fluctuates with respect to the deforestation experienced by that location at different spatial lags. For example, a small window with high Pn embedded within a large window with low Pn implies a locally intensive deforestation (content) nested in a larger region with less forest loss (context). Therefore, the resultant profiles describe deforestation dynamics from local to regional scale (here, from 0.0225 km^2 to 259.5321 km^2), and could therefore be incorporated into conservation programs to monitor system dynamics across multiple spatial scales.

In Figures 4-3, 4-4, 4-5, the asymptotic behaviours of the five clusters of all three states suggest the scale-dependent pattern of percentage developed areas was adequately described with the multiple window sizes used, ranging two magnitudes from 0.15 km to 16.11 km. Though the exact cluster means are different, the relative

trend of these profiles behaved similarly. The core developed areas typically have higher densities at local scales (small windows) and then decrease as the window size increases, so do the sub-developed areas. Both the core forest areas and the sub-forest areas have the converse pattern, that is, their densities increase with window size increases, where the sub-forest areas increase more rapidly than the core forest areas. The transition areas have an interesting profile, the densities increase quickly at relatively small window sizes and then maintain at a relatively stable state. In sum, the profiles described characterize the fact that intensive fragmentation occurs locally in the MAP region, while a large part of this region is still covered by forest.

Focus area study: Rio Branco

To carefully examine the 'halo' impacts of deforestation, a region around Rio Branco, the capital city of Acre, was subset as an illustration (Figure 4-6). Rio Branco was one of the first settlements to appear on the banks of the Acre River, and has become the administrative centre of the economic and cultural region.

According to Table 4-2, the non-forest area increases from 4.71% in 1986 to 34.80% in 2010 (where 1% in Acre represents 622.79 km^2) and consequently, the forest area shrinks from 95.29% in 1986 to 65.20% in 2010, where 65.20% is close to a critical percolation threshold of 59.28% as proposed by Gardner (1999) with any value below this threshold representing an evident loss of landscape connectivity. The most recent demographic census reported that the population in Rio Branco was 305,954 in 2009 (Brazilian Institute of Geography and Statistics). According to the definition of the US census, urbanized areas are defined as densely settled territories that contain 50,000 or more people. In Table 4-3, the increase of the core developed area around Rio Branco, including major towns and small settlements, can be re-defined as urban expansion

which accounts for 3.3% of the subset area in 1986 and increases to 24.6% in 2010. The impacts of urban expansion, including suburbanization area (i.e. sub-developed area), transition area, and sub-forest area, grows from 40.1% in 1986 (5.6% + 8.8% + 25.7% = 40.1%) to 47.74 % in 2010 (12.2% + 11.1% +24.5% = 47.8 %). The 'halo' impacts stretch over large areas that are almost twice that of the actual urban fabric and this confirms that urbanization exhibits strong influences beyond the extent of its boundaries, and thus this improved across-scale analysis works better than the traditional methods that present forest/non-forest only.

Summary

Window analysis is a widely used technique in forest fragmentation analyses, as users can define various convolution kernels to capture composition and configuration information as needed (e.g. Riitters et al., 2000, 2002). Convolution kernels have regular geometric shapes, such as rectangle, circle, hexagon, while fragmented areas or patches in the landscape are not regular and such mismatch between the boundary of window and target object may limit the applications. To address this boundary problem, morphological spatial pattern analysis was proposed to map forest morphology, such as corridors, scale pattern, and connectivity (MSPA, Soille, 2003; Vogt et al., 2007a, 2007b). Specifically, MSPA is used to solve the problem of ambiguity between forest exterior boundary (edge) and interior fragmentation boundary (perforation) and therefore, as a pixel level analysis MSPA can be used as a complementary reference of the window analysis that prompts the understanding of deforestation and its related problems.

There is no single correct scale or level at which to describe a system, nor does that mean all scales serve equally well (Levin, 1992). In the spatial domain, grain and

extent are inversely correlated, where the detailed information of fine grain would be inevitably sacrificed at large scales. A heterogeneous landscape at a relatively small scale may become homogenous at a large scale, and vice versa. The moving window technique with different window sizes provides an approach to describe such ‘across scales’ information, however various spatial patterns analysed from multiple window sizes may lead to different statistical relationships that are not able to provide consistent references. To overcome such problems, the unsupervised ISO algorithm was used here to integrate various spatial patterns analysed from different window sizes and then expresses such information by profiles ranging from local to regional scales, as such, this integrated ‘across scale’ analysis by ISO algorithm helps reveal the phenomena under study or the process behind it over an appropriate range of scales. In this paper, the smallest scale of the profile is 0.15 km and such fine resolution would be able to provide a practical guide for conservation programs, and the largest scale of the profile is 16.11 km, where the spatial patterns analysed by such large windows provides an understanding of deforestation at a regional level. Additional window sizes can be added to calculate the information at higher levels, and the largest size should be less than the entire landscape.

In recent decades, initialized by the road paving and then enhanced by the urbanization, forest landscapes in the MAP region has been largely affected by rapid land cover change, such that with respect to many social and ecological concerns, the mapping and interpretation of forest cover changes are needed. However, due to spatial autocorrelation (or the first law of Geography), the forest security of the deforested landscape is not uniformly distributed, but rather, exhibits a gradient that decreases

from the interior forest to the interface between forest areas and cleared areas. Such gradients make ‘halo’ mapping necessary to complement the traditional forest/non-forest mapping. The ‘halo’ impact is derived from the composition of forest within the moving window and has actually been used as a pattern (configuration) indicator in the paper. Based on such indicators, the five calculated clusters can be used to characterize the complex interaction between forest and non-forest. The increase of the core developed area represents the need to satisfy the growth of urban populations, and the increase of the sub-developed area is used to accommodate more urban populations or is being modified to create more agricultural area to feed the larger numbers of people. And both the core developed and the sub-developed areas are closely related to the urban growth, social and economic developments. The transition area is the most unstable cluster identified, as for the most part either deforestation or reforestation would be occurring here, so the transition area becomes a hotspot that should be considered as an ideal area to measure biodiversity loss and/or urban sprawl. The sub-forest area is the least affected cluster by the ‘halo’ impacts of the developed area, and its change could indicate a very early signal of either deforestation or reforestation. Finally, the core forest represents the most stable forest distribution and therefore any substantial reduction of this cluster would signal a fundamental transition of the forest area as well as the entire forest system.

The accumulated time-series forest cover maps from 1986 to 2010 (Cumming et al., 2005, Cumming et al., 2008, Southworth et al., 2011, Marsik et al., 2011, Cumming et al., 2012) reveal that the MAP region is still largely forested, but a fundamental shift in forest landscape is underway among the three regions based on this ‘halo’ mapping

analysis. The fluctuation of these five clusters along both spatial and temporal scales, and particularly the different dynamics among the three regions, requires policy regulation to be adaptive to different clusters and be flexible for their evolving properties, as well as acting in a timely manner. Hence, such mapping may have an influence in either ecological conservation or economic planning, depending on government policies, either addressing sustainable environment or economic growth.

The cartographic representation of the ‘halo’ impact of deforestation is based on the combination of GIS/remote sensing techniques and spatial statistics, and such integration would be useful to provide a framework to study the influences of land use and land cover problems that are not restricted by visible borders. The method used here is applicable and has been verified by a deforested example in the Amazon and hence, such a simple framework should be widely adopted to implement the automatic classification of deforested landscapes, as well be used to monitor and address forest/urban regulation and sustainability.

Table 4-1. Summary of clusters' areas in percentage from 1986 to 2010 in the MAP region*

Region	Cluster	1986 (%)	1991(%)	1996(%)	2000(%)	2005(%)	2010(%)
Acre	Core Developed	1.7	3.4	4.0	5.6	9.0	13.3
	Sub-Developed	3.0	4.2	4.4	5.8	6.7	6.7
	Transition	4.3	3.9	5.6	5.7	6.6	6.7
	Sub-Forest	16.0	16.5	16.9	16.1	15.2	14.7
	Core Forest	75.0	72.0	69.3	66.8	62.5	58.6
Madre de Dios	Core Developed	1.5	1.1	0.5	0.9	0.9	1.7
	Sub-Developed	2.5	2.0	1.1	2.1	1.9	2.5
	Transition	3.5	2.6	1.2	2.1	2.6	2.5
	Sub-Forest	17.4	16.8	10.2	20.1	19.4	17.3
	Core Forest	75.1	77.5	87.0	74.2	75.2	75.7
Pando	Core Developed	0.2	0.3	0.4	0.8	1.0	2.0
	Sub-Developed	0.5	0.5	0.7	1.4	1.6	2.5
	Transition	0.6	3.1	0.8	1.0	1.3	3.3
	Sub-Forest	5.5	3.7	8.5	10.4	11.2	18.5
	Core Forest	93.2	92.4	89.6	86.5	84.9	73.7

* where 1% in Acre represents 622.79 km^2, 1% in Madre de Dios represents 493.48km^2, and 1% in Pando represents 542.87 km^2.

Table 4-2. Summary of forest/non-forest areas from 1986 to 2010 around Rio Branco, Acre. *

Region	Cluster	1986 (%)	1991(%)	1996(%)	2000(%)	2005(%)	2010(%)
Rio Branco	Forest	95.3	88.5	85.0	82.9	73.2	65.2
	Non-forest	4.7	11.5	15.0	17.1	26.8	34.8

* where 1% in Acre represents 622.79 km^2.

Table 4-3. Summary of clusters' areas in percentage from 1986 to 2010 around Rio Branco, Acre. *

Region	Cluster	1986 (%)	1991(%)	1996(%)	2000(%)	2005(%)	2010(%)
Around Rio Branco	Core Developed	3.3	8.3	9.9	11.7	19.0	24.6
	Sub-Developed	5.6	5.3	7.2	8.6	10.4	12.2
	Transition	8.8	8.5	11.4	10.8	11.9	11.1
	Sub-Forest	25.7	26.1	28.8	28.1	26.9	24.5
	Core Forest	56.6	51.8	42.7	40.8	31.8	27.6

* where 1% in Acre represents 622.79 km^2.

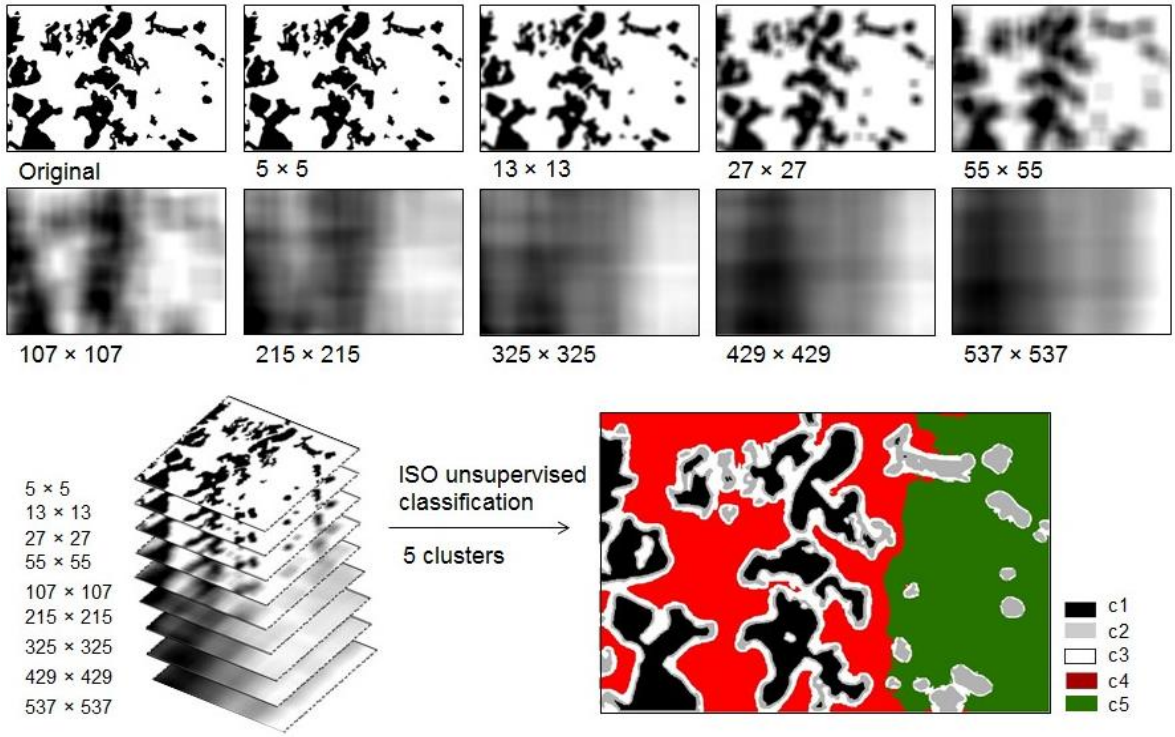

Figure 4-1. A simulated landscape (spatial resolution: 30m) was convoluted by 9 different filters, ranging from 5×5 to 537 ×537, and these such 9 filtered maps were then stacked to produce a multi-band layer stack image and to then run an ISO classification with user-defined cluster numbers (5 clusters here from c1 to c5).

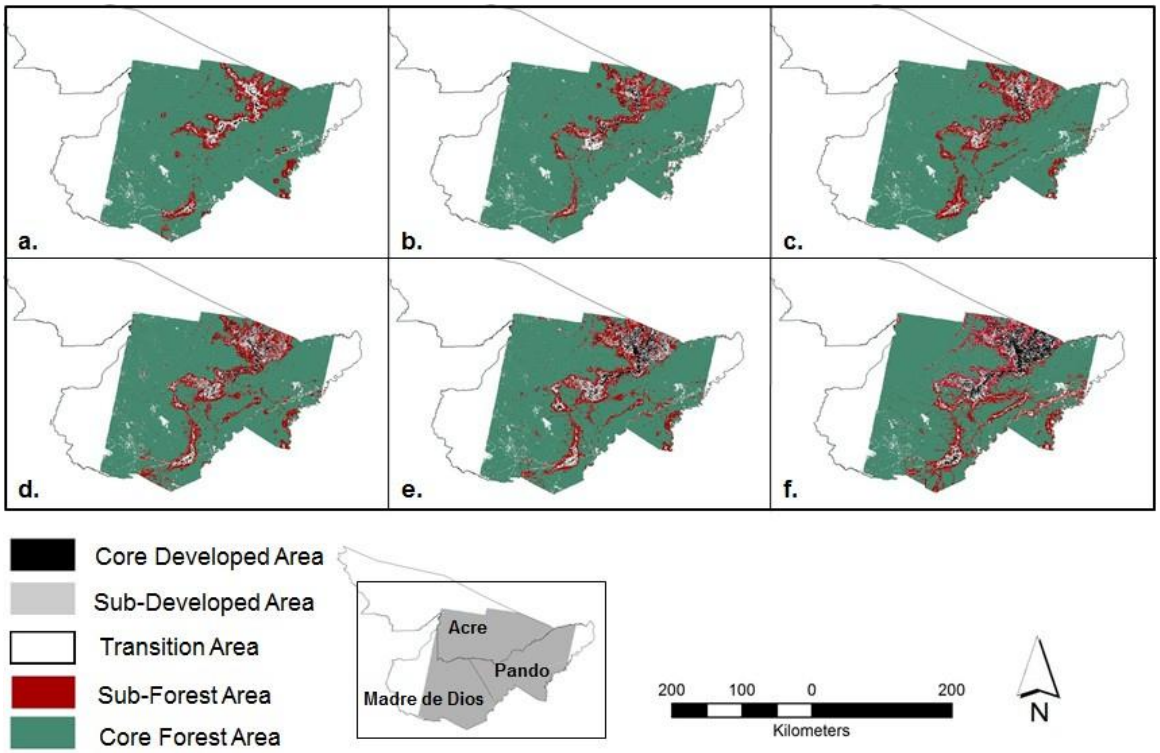

Figure 4-2. The results of the clustering analysis in the MAP region from 1986 to 2010, using the migrating means clustering algorithm (also known as the ISO cluster) which is based on the similarity of P_n (fragmentation proportion) within 9 window sizes. A) 1986. B) 1991.C) 1996. D) 2000. E) 2005. F) 2010.

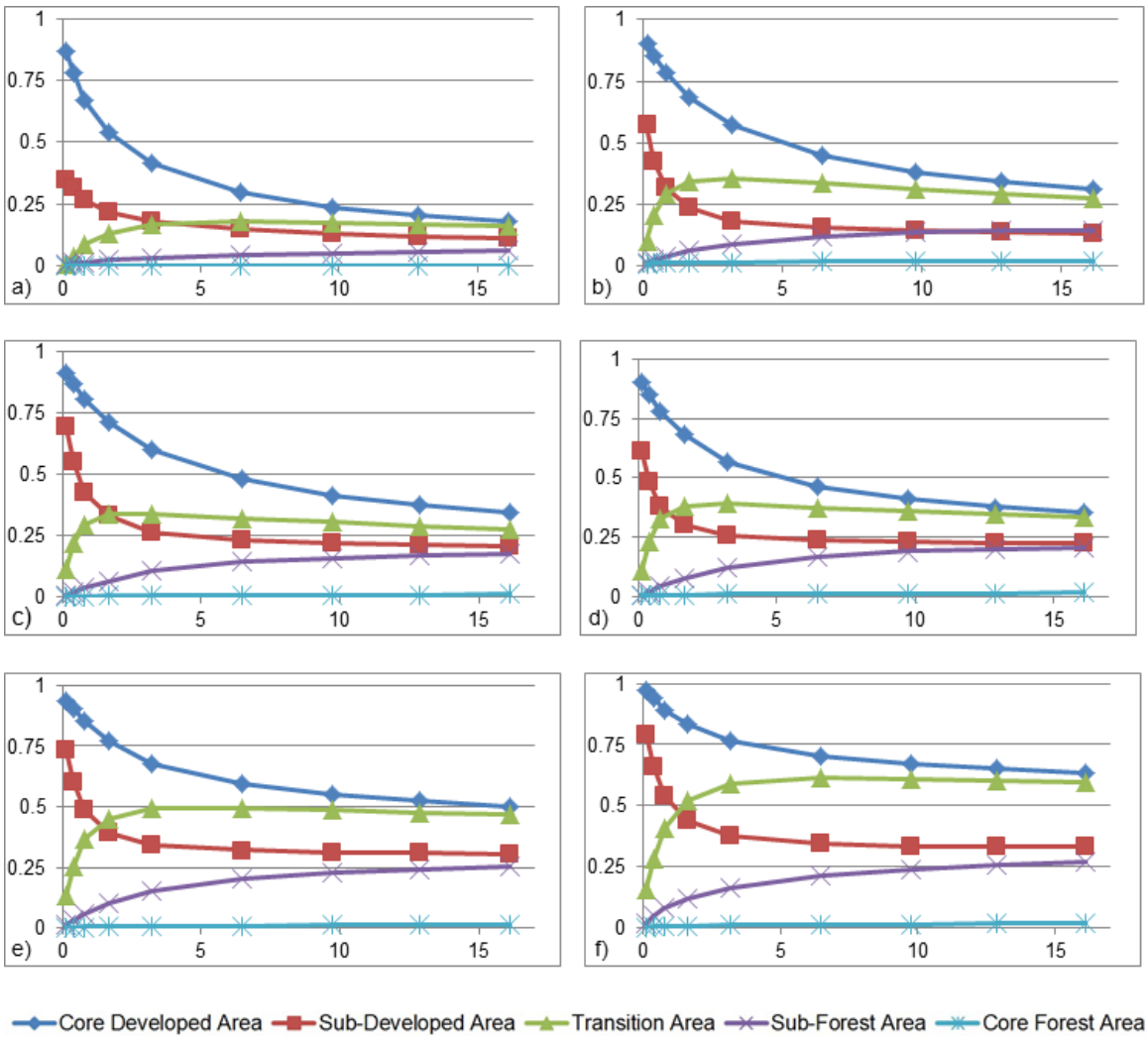

Figure 4-3. Cluster means at all scales in Acre, Brazil from 1986 to 2010. X-axis is the side length of different windows (unit: km) and Y-axis is P_n (fragmentation proportion within each window) A) 1986. B) 1991.C) 1996. D) 2000. E) 2005. F) 2010.

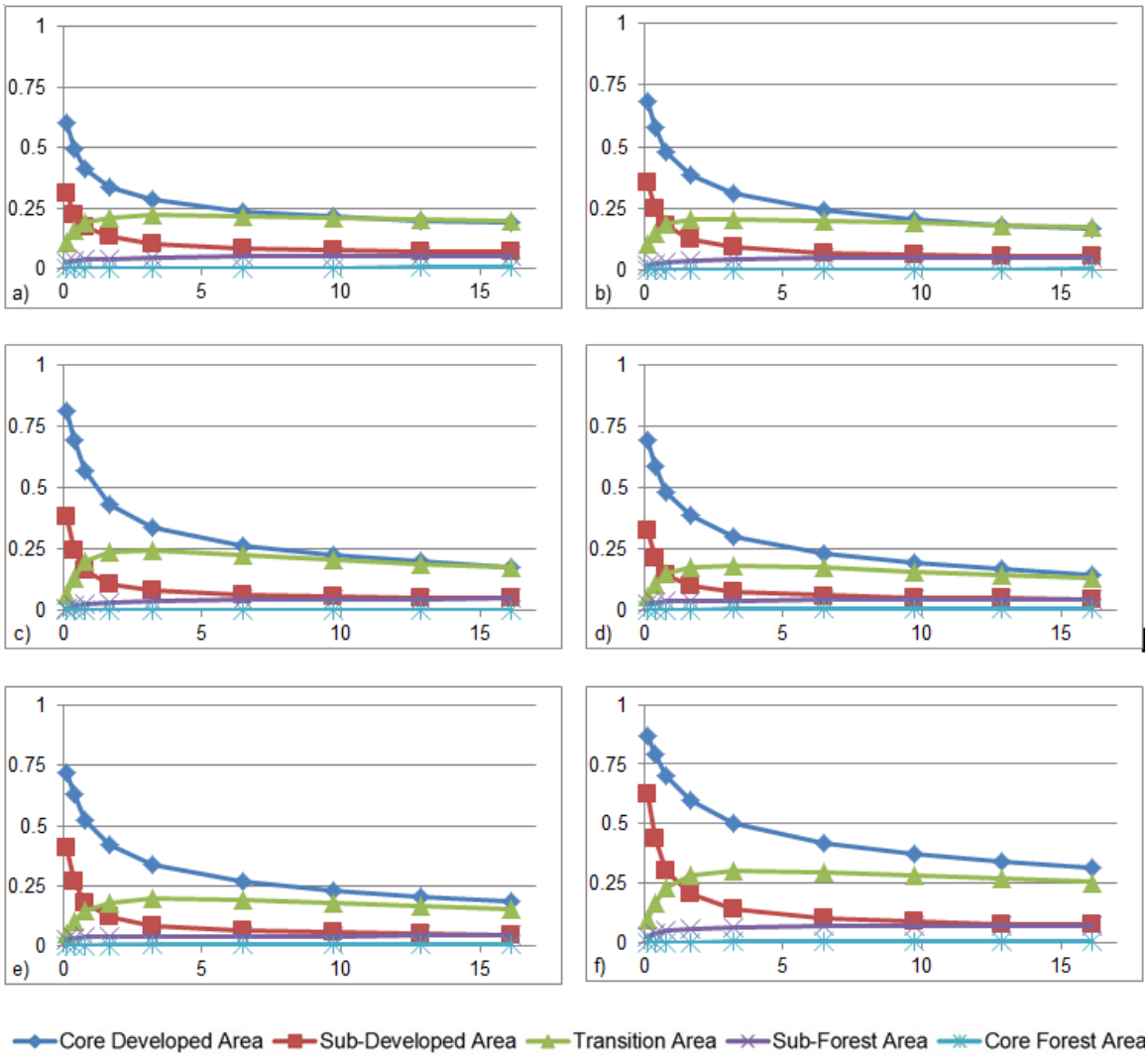

Figure 4-4. Cluster means at all scales in Pando, Boliva from 1986 to 2010. X-axis is the side length of different windows (unit: km) and Y-axis is P_n (fragmentation proportion within each window) A) 1986. B) 1991.C) 1996. D) 2000. E) 2005. F) 2010.

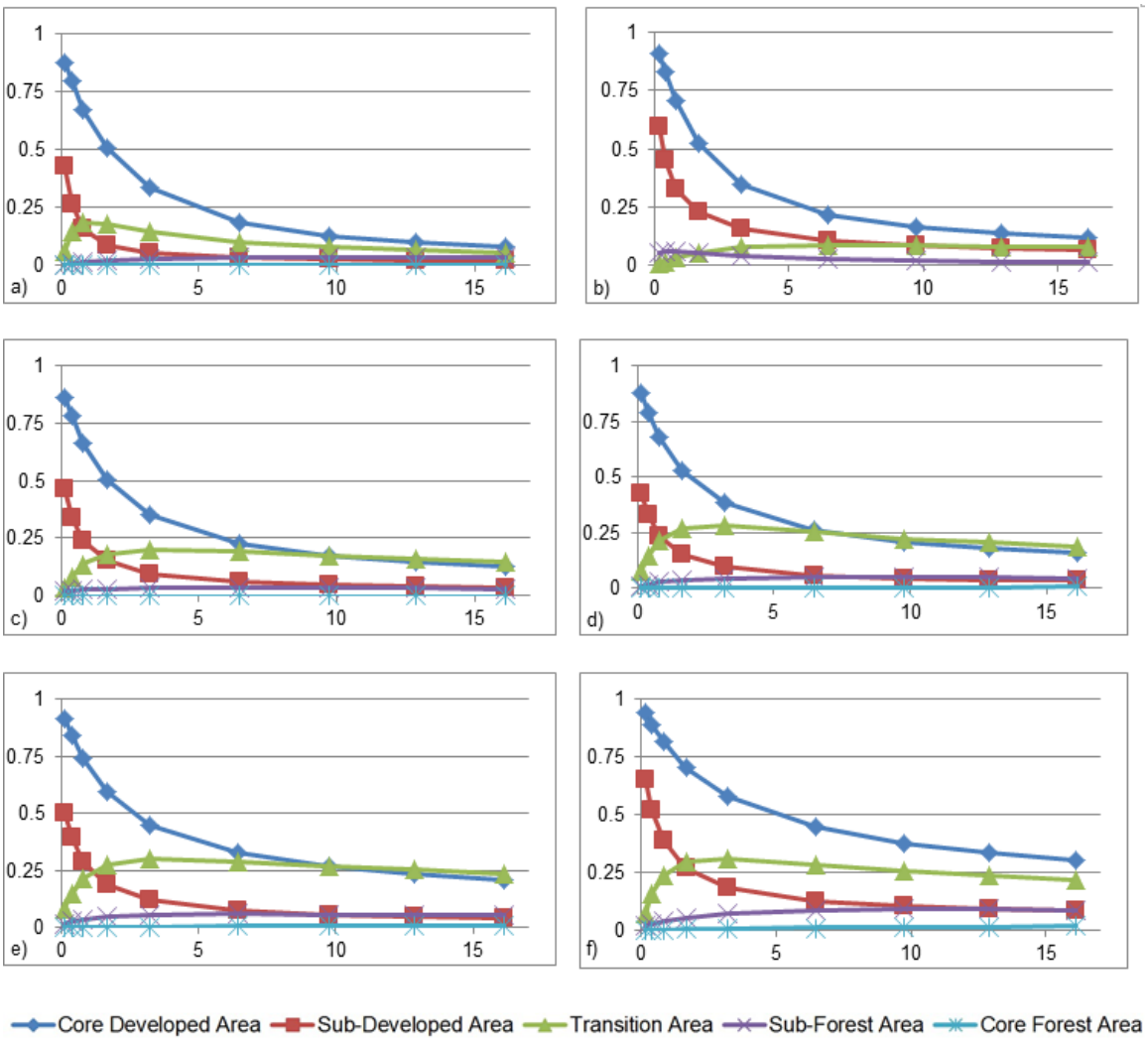

Figure 4-5. Cluster means at all scales in Madre de Dios, Peru from 1986 to 2010. X-axis is the side length of different windows (unit: km) and Y-axis is P_n (fragmentation proportion within each window) A) 1986. B) 1991.C) 1996. D) 2000. E) 2005. F) 2010.

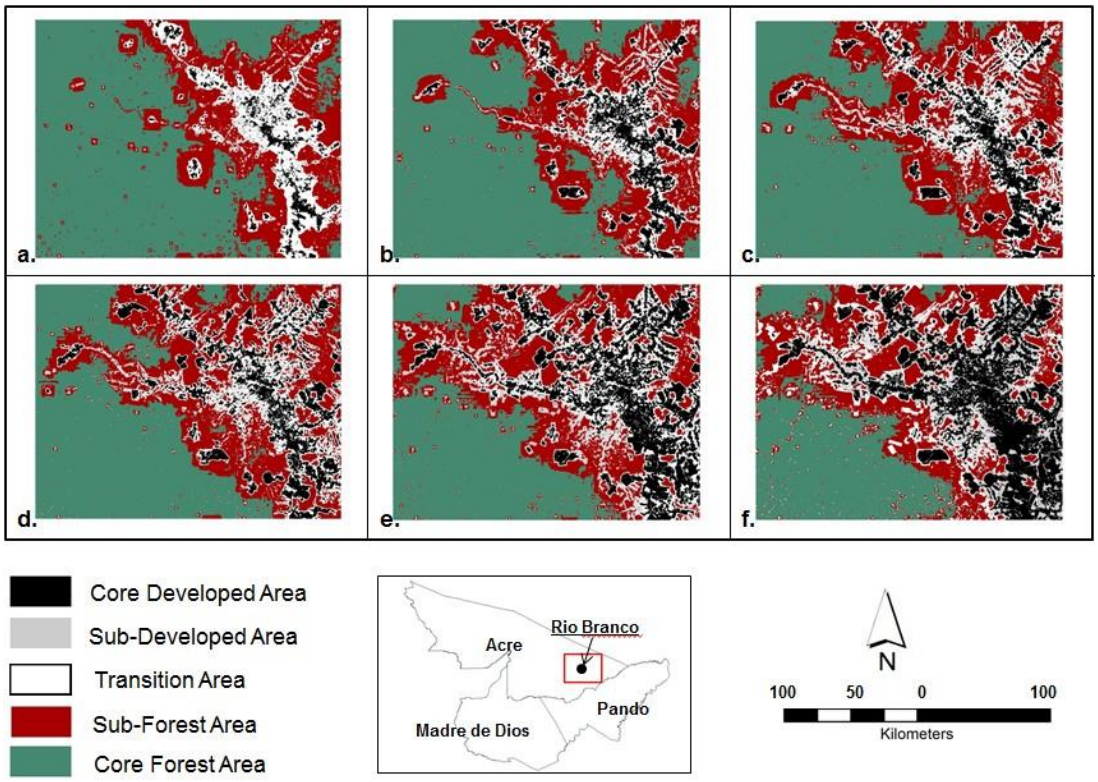

Figure 4-6. The results of the clustering analysis around Rio Branco, Capital of Acre, Brazil, from 1986 to 2010, using the migrating means clustering algorithm (also known as the ISO cluster) which is based on the similarity of P_n (fragmentation proportion) within 9 window sizes. A) 1986. B) 1991.C) 1996. D) 2000. E) 2005. F) 2010.

CHAPTER 5
INDICATING STRUCTURAL CONNECTIVITY USING MORPHOLOGICAL IMAGE PROCESSING ANALYSIS*

Background

Large-scale deforestation studies became feasible since 1972 with the analysis of satellite imagery (Giles and Burgoyne, 2008), and there is a large collection of compelling examples of deforestation studies (Nepstad et al., 1999, 2001; Southworth and Tucker, 2001; Laurance et al., 2001, 2002; Brook et al., 2003; Nagendra, 2004). When viewing a landscape, we look at its composition and spatial configuration (Li and Reynolds, 1994), that is, both the elements present and how these elements are arranged (Turner et al., 2001; Wu and Hobbs, 2002). Correspondingly, forest fragmentation refers to two processes – the first is the area of forest cleared and the second is the pattern of this clearing. Both of these factors are important for a variety of ecological reasons – species survivability, species dispersal, species migration patterns etc. Deforestation affects the spatial layout of all components of the forest system. Many satellite-based tropical surveys have treated deforestation as a temporal change in forest extent, and not as a spatial property of forests (Foody and Curran, 1994). A given amount of forest can be arranged in many patterns (Riitters et al., 2002), and spatial patterns have been shown to influence many processes that are ecologically important (Turner, 1989). Substantially different spatial arrangements with the same abundance of forest cover (Vogt et al. 2007a, 2007b) could have totally different ecological functions. Therefore, quantitative methods are required to detect spatial patterns of deforestation,

* "Reprinted with permission from: Sun, J., & Southworth, J. Indicating structural connectivity in Amazonian Rainforests from 1986 to 2010 using morphological image processing analysis, *International Journal of Remote Sensing*. 34 (14): 5187-5200."

identify significant changes through time, and relate these patterns to important ecological functions.

Based on mathematical morphology theory (Soille, 2003), morphological spatial pattern analysis (MSPA) identifies spatial patterns from a single-layer binary map, and describes the size, shape and other geometric characteristics of spatial entities. For a forest/non-forest map, the existing MSPA algorithm focuses on the spatial information of forest areas without giving enough consideration to non-forest areas. The spatial distributions of non-forest areas should be emphasized, because the perforated fragmentation inside the continuous forest and the exterior fragmentation on forest boundaries would exhibit different influences on landscape connectivity. During the deforestation process, with the same amount of forest loss, the increased perforated fragmentation may split a continuous forest into discrete patches, while exterior fragmentation may only remove forest areas from the frontier but still maintain a contiguous forest area. It follows logically that widely connected perforated fragmentation has a stronger early warning signal on connectivity loss than that of exterior fragmentation. Therefore, to better understand the impacts on connectivity from both forest and non-forest areas, it is necessary to further identify non-forest areas and then add the new patterns to the existing MSPA classes.

Chapter 5 aims to assess forest/non-forest changes and examine how these spatial patterns change and how these changes themselves influence landscape connectivity. Such quantitative mapping of forest/non-forest patterns at a pixel-level would be able to support forest condition evaluation and long-term conservation programs in the Amazon.

Morphological Spatial Pattern Analysis

Existing MSPA

MSPA uses a series of image processing routines to identify different spatial patterns at a pixel-level. The value of each forest pixel in the output image is based on the connectivity with its neighbors. Connectivity can be set to either four (pixels are connected if their edges touch) or eight neighbors (pixels are connected if their edges or corners touch). Dilation and erosion are two fundamental morphological operations in MSPA that traverse the whole image. Based on connectivity, dilation adds pixels to the boundaries of objects in an image, while erosion removes pixels on object boundaries (Soille 2003). The pattern classes identified by MSPA include core, islet, bridge, loop, branch, edge, and perforation, which are listed in Table 5-1 (MSPA column) and illustrated in Figure 5-1 (Soille and Vogt, 2009). To stress the advantages that can only be achieved by MSPA, we compare the MSPA classes with the commonly used landscape metrics provided by Fragstats (McGarigal et al., 2002) in Table 5-1. The MSPA operations described are implemented by the software package GUIDOS (version 1.3)

Proposed additional MSPA classes: hole and outer background

The existing seven MSPA classes described previously focus on the configuration of foreground pixels, that is, the spatial arrangement of forest areas, where the background pixels, e.g. non-forest areas were simply treated as a single class. MSPA was first proposed specifically to solve the problem of ambiguity between edge and perforation (Soille, 2003; Vogt et al., 2007a, 2007b), which corresponds to the outer boundaries and inner boundaries. Perforation is the result of the process of making holes in an area, or rather, the interruption of land cover continuity by the

formation of openings or gaps in canopy cover (Bogaert et al., 2004). A hole is defined as a connected component of the background that does not contain any pixel of the border of the image, where perforation pixels of a given connected component (patch) are defined as its boundaries that contact with the hole (Soille and Vogt, 2009). Thus, perforation pixels of a given connected component (patch) are defined as its boundary pixels that are within certain distance to a hole of this connected component where a hole is defined as a connected component of the background that does not contain any pixel of the border of the image (Soille and Vogt, 2009). Perforations introduce fragmentation impacts deeper into the forest, in comparison to removing the same amount of forest on the exterior boundary of a large forest tract (Riitters et al., 2002). The hole is delineated by perforation pixels, but perforation pixels (object perimeter) and hole pixels (object area) are not a 1:1 relationship and thus using only the perforation feature we cannot provide the information regarding hole extent. Given such perforated holes exhibit rapid impacts to fragmentation of the connected forest by breaking it into discrete patches this is of paramount importance to both measure and understand. In this respect, we propose that background pixels (non-forest class) could be further separated into ‘hole’ and ‘outer background’ as illustrated in Figure 5-2, where the hole is delineated by perforation pixels, and outer background is delineated by edge pixels (outer boundaries). In this way we expand upon the use of MSPA in fragmentation and conservation studies by focusing not only on the dominant landscape features (here set as forest) but also on the background matrix (here non-forest). Here we have divided the ‘background’ features into ‘hole’ (internal background non-forest cover found within the matrix of forest cover) and ‘outer background’ features (external background non-

forest cover found on perimeter of existing forest regions). As such we now locate our 'background' (non-forest) class on the landscape, and specifically locate it within the forest matrix thus allowing us to better understand our landscape.

MSPA Results

We applied MSPA to the three states of the MAP region and set MSPA connectivity rules to the eight-neighbor with one pixel as edge width, where we hypothesized that the eight-neighbor rule is closer to the real world representation of neighbor influences in this landscape than a four-neighbor rule. The MSPA results (without the two newly proposed background classes) are displayed in Table 5-2. Over the past 25 years, we observed an increase in non-forest cover for all three states (Acre, Madre de Dios, and Pando), at the expense of forest areas. Overall, our results across all three regions reveal a landscape still dominated by forest cover up through 2010 (Acre – 80.0%, Madre de Dios – 97.40%, and Pando – 98.02%). There are different trends in the various morphological statistics across the three states, and the different morphological groups will be discussed below, in terms of percentage of state area in each class.

Core and Boundary: Edge and Perforation

Core represents the interior area of a forest patch. For Acre (95.69 % in 1986 to 74.86% in 2010) and Pando (99.21% in 1986 to 95.21% in 2010), the core area has declined across the study period. For Madre de Dios a peak in forest core area was observed in 1996 (97.49%), but overall core area has increased from 92.56% in 1986 to 95.01% in 2010.

For both Acre and Pando the increase in edge area is simultaneous with the decline in core area. Interestingly, Madre de Dios follows this same pattern up to 2010

when, despite an increase in forest core area, there is an increase in edge area. In Acre, the area representing the perforated pixels was at a maximum in 1991 (3.89%) and has declined from 1991 through 2005 (1.19%) until 2010 where this value increased slightly to 1.97%. In Pando, with much lower rates of perforation (and forest clearing) overall, 1986 has only 0.24% perforation and this stays below 1% until 2010, where an increase to 1.56% perforation and an increase in edge (from 0.03% in 1986 to 0.52% in 2010) together illustrate a beginning or initialization of the opening up of this predominantly forested landscape. Madre de Dios has a perforation rate of 2.26% in 1986, this drop to 1% by 1996 and then increases to around 2.53% in 2000 and 2.45% in 2005. This declines once more to around 1.07% by 2010, although this same period sees a slight decrease in edge.

Islet

Islets are areas of forest cover which are not connected to any core areas. Islet pixels are extremely low for all three states, Pando (0.01% in 1986 to 0.05% in 2010), Madre de Dios (around 0.04%), and Acre (0.05% in 1986 to 0.22% in 2010).

Connector: Bridge, Loop, and Branch

Connector pixels within our landscape are here designated as bridge, loop, and branch. For Pando the presence of loop (0.07 in 1986 to 0.32% in 2010), bridge (0.02 in 1986 to 0.17% in 2010) and branch (0.02 in 1986 to 0.19% in 2010) remains very low although these values do increase from the minimum in 1986 to their maximums in 2010. For Madre de Dios the lowest values of loop, bridge and branch features occur in 1996 (0.25%, 0.09%, and 0.06%, respectively), higher values occur in 1986, 2000 and 2005. By 2010 the values are still low (0.36%, 0.20%, and0.15%, respectively) again illustrating these really are quite minor landscape features at this time. Finally Acre,

which is much further along in the forest fragmentation process than Pando or Madre de Dios, has higher values in general when compared to the other states but these are still quite low overall (loop – 0.22% in 1986 to 0. 45% in 2010, bridge – 0.12% in 1986 to 0.63% in 2010, and branch– 0.18% in 1986 to 0.47% in 2010).

Background: Hole and Outer background

In a perforated landscape, a large (relative to landscape size) and non-compact forest cluster typically has “holes” created by a small amount of non-forest land cover (Riitters et al. 2002). To understand this ‘gap formation’ process, we summarized the statistic results of hole areas as well as outer background areas in Table 5-3. The hole pixels experienced an increase from 2.23 % in 1986 till 2.96% in 2000, and then decreased to 1.85% in 2010. For both Madre de Dios and Pando, their hole profiles behave similarly: increasing from 1986 (Madre de Dios 0.81%, Pando 0.35%) to 2010 (Madre de Dios 0.94%, Pando 1.37%), with minimal values occurring in 1996 (Madre de Dios 0.47%, Pando 0.52%). The outer background in Acre has been increasing from 1986 (0.31%) to 2010 (18.12%), while in both Madre de Dios (1.34% in 1986 and 1.66% in 2010 with lowest value 0.44% occurred in 1996) and Pando (0.05% in 1986 and 0.61% in 2010), the profiles of outer background are relatively stable.

A typical inverse U-shape profile of hole class was observed in Acre. When the total forest cover decreased monotonically over the 25 years in Acre, this U-shape profile could indicate that discrete fragmentations first occurred inside the continuous forest at a small scale, and as the deforesting areas continued increasing, some holes started to connect with neighbor holes. During this process, the hole areas had grown large enough to break through forest outer boundaries (edge class here), and thus these holes transformed into outer background. Thus, it is naturally to conclude that the

spatial information of background pixels needs to be emphasized and classified, as hole and outer background exhibit different impacts during the deforestation process. Specifically, the hole class represents the fragmentation occurs inside forest and its typical inverse U-shape profile described reveals a deforestation process: from a dispersed fragmented area to a highly discrete area.

Discussion

Scale problems in MSPA applications

It is noteworthy that Figures 5-1 and 5-2 are illustrations where we have focused on a very small sub-region, at a pixel level as MSPA is constrained by image resolution. On the contrary, the study here is a real landscape and thus may cause misinterpretations of MSPA classes. Among the defined MSPA classes, islet pixels and connector pixels, including bridge, loop, and branch may not be able to serve any ecological function. Islet class could be understood as small patch; however, they are of little value here when looking at a landscape level. With a spatial resolution of 30 meter, an islet would simply be a very few number of trees remaining isolated in the landscape. The almost negligible proportion of loop, bridge and branch pixels may indicate that deforestation is not selective but happening in large or more extensive areas. In any case, the very low number of these pixels shows that connectivity is essentially provided through the core areas. Connector pixels alone are not representative for connectivity. For example, a connector may widen over time and consequently turn into a core area; here the connector pixels decreased or were even lost completely while connectivity has increased. Forest regrowth may result in loop and branch features disappearing but connectivity still increasing.

Inferences: A conceptual connectivity model

In contrast to traditional fragmentation analyses (Southworth et al., 2001, 2006; Riitters et al., 2000, 2002; Nagendra et al., 2003, Marsik et al., 2011), the research approach taken here is based on mathematical morphological methods (Soille, 2003, Soille and Vogt, 2009) and provides simultaneous input on the spatially explicit information regarding the landscapes configuration (i.e. pattern) and composition (i.e., area and number of fragments). The use of mathematical morphologies provides an excellent suite of meaningful and interpretable indices as illustrated in this research.

MSPA results provide concrete data to infer connectivity dynamics in the MAP region, and we summed up this inference graphically with a conceptual model (Figure 5-3). In this model, a deforested landscape experiences three stages from the 'onset' stage via a 'transition' stage to a 'final' stage, where the forest landscape loses its connectivity completely due to decreased forest percentage. At the beginning of this deforestation process, the landscape is dominated by the core class, where the forest is closely connected. When the total forest percentage continues decreasing, the hole, perforation, and edge classes are dominant, as fragmentation causes more forest to be exposed to non-forest surfaces. Such forest landscape is still connected though forest function and service are now being lost. From the transition stage to the final stage, the landscape has changed qualitatively from predominantly forest area to non-forest area due to extremely low forest cover, where loop, bridge, branch, and islet are the dominant classes left, and most non-forest areas are outer non-forest. At this final stage a landscape is dominated by non-forest cover and so as such the forested landscape would be considered lacking in connectivity. Our results reveal that all three regions are still relatively early in their landscape transitions, and based on the statistical summary

(Figure 5-1, Table 5-2), the conceptual model indicates that both Madre de Dios and Pando are in the onset stage, while Acre is transforming into the transition stage (Figure 5-3).

Application: A deforested landscape in the Amazon

MSPA has been quite widely used for such uses as mapping spatial patterns, landscape corridors, scale pattern, landscape functional connectivity, and to assess green infrastructure (Vogt et al., 2007a, 2007b, 2009; Riitters et al., 2007, Ostapowicz et al., 2008, Wickham et al., 2010). In both the Europe and the US, forest maps have been morphologically classified to indicate forest ecosystem quality and to assess the amount of fragmentation present in a landscape as well its potential impact.

The Amazon contains about half of the earth's remaining tropical forests, where deforestation is growing rapidly and will likely continue to do so in the future, and this in turn, results in the development of a substantial body of publications. However, fine or medium resolution land cover products are not available for South America, especially at the scale of the whole Amazon, and consequently, there is no spatial pattern analysis that extends to such a large scope. Due to continuous work in the MAP region (Cumming et al., 2005, Cumming et al., 2008, Southworth et al., 2011, Marsik et al., 2011, Cumming et al., 2012), we have accumulated a time-series of forest/non-forest maps from 1986 through 2010. Thanks to this set of data, we can interpret the indications of different MSPA classes from their profiles, and based on the results, build a conceptual model to represent landscape connectivity. Moreover, this paper enriches the paucity of MSPA applications in the Amazon, and the "split" of hole from background emphasizes that the spatial properties of background are as important as those of

foreground (forest pixels), such that should be included into conservation programs to promote the understanding of forest dynamic complementarily.

Table 5-1. Fragstats indices compared to the MSPA class metrics.

MSPA Class	MSPA	Fragstats
Edge	External perimeter of core area	Perimeter of core area
Islet	Small patch that is too small to contain a core area.	No difference between patch size, all patches are treated the same
Branch	A state between connected and disconnected. A degraded corridor in deforestation, or a construction corridor in reforestation	No similar index
Loop	Fragmentation that happened next to the forest edge	No similar index
Bridge	Connection to different core areas.	Patch cohesion (COHESION) is computed from the information contained in patch area and perimeter, which is the proportional to the area-weighted mean perimeter-area ratio divided by the area-weighted mean patch shape index. Connectance (CONNECT) is defined on the number of functional joinings, where each pair of patches is either connected or not based on some criterions, and FRAGSTATS computes connectance using a threshold distance specified by the user and reports it as a percentage of the maximum possible connectance given the number of patches. Also, traversability (TRAVERSE), based on the idea of ecological resistance, can measure the connectivity. Besides the Fragstats, there are some papers discuss landscape connectivity and build indices to measure it.
Perforation	Internal perimeter of core area.	There is no index to represent perforation, which occurs inside the forest. Some indices combination can indicate fragmentation, such as, total edge (TE), edge density (ED), and landscape similarity index (LSI) can measure the fragmentation.
Core	Interior area determined by edge width	Interior area determined by the edge width

Table 5-2. Statistical summary of MSPA classes of the three sides of the MAP region from 1986 to 2010. *

Region	Class	1986(%)	1991 (%)	1996 (%)	2000(%)	2005(%)	2010(%)
Acre	Developed	2.5	6.6	7.5	9.6	14.4	20.0
	Core	95.7	86.8	88.0	84.7	80.9	74.9
	Edge	0.2	0.8	1.0	1.4	1.7	1.8
	Islet	0.1	0.1	0.2	0.3	0.3	0.2
	Branch	0.2	0.4	0.5	0.6	0.6	0.5
	Loop	0.2	1.1	0.5	0.6	0.4	0.5
	Bridge	0.1	0.3	0.3	0.6	0.4	0.3
	Perforation	1.0	3.9	2.1	2.1	1.2	2.0
Madre de Dios	Developed	2.2	1.7	1.0	1.8	2.0	2.6
	Core	92.6	95.1	97.5	93.6	93.6	95.0
	Edge	0.9	0.6	0.2	0.5	0.5	0.6
	Islet	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Branch	0.3	0.2	0.1	0.2	0.2	0.2
	Loop	0.8	0.5	0.3	0.8	0.8	0.4
	Bridge	1.1	0.6	0.1	0.6	0.5	0.2
	Perforation	2.3	1.3	1.0	2.5	2.5	1.1
Pando	Developed	0.4	0.7	0.8	1.3	1.9	1.9
	Core	99.2	98.6	98.0	97.5	96.5	95.2
	Edge	<0.1	0.1	0.1	0.1	0.2	0.5
	Islet	<0.1	<0.1	<0.1	<0.1	<0.1	0.1
	Branch	<0.1	<0.1	0.1	0.1	0.1	0.2
	Loop	0.1	0.1	0.2	0.2	0.3	0.3
	Bridge	<0.1	<0.1	0.1	0.1	0.1	0.2
	Perforation	0.2	0.4	0.8	0.6	0.9	1.6

*where the total area for Acre is about 62279km^2, the total area for Madre de Dios is about 49348 km^2, the total area for Pando is about 54287 km^2.

Table 5-3. Statistical summary of additional MSPA classes ‘hole’ and ‘outer background’ over three states in the MAP region from 1986 to 2010. *

Region	Class	1986(%)	1991 (%)	1996 (%)	2000(%)	2005(%)	2010(%)
Acre	Hole	2.2	2.9	2.8	3.0	2.1	1.9
	Outer background	0.3	3.7	4.7	6.7	12.3	18.1
Madre de Dios	Hole	0.8	0.8	0.5	1.2	1.2	1.0
	Outer background	1.3	0.9	0.4	0.6	0.9	1.7
Pando	Hole	0.4	0.5	0.5	1.0	1.4	1.4
	Outer background	0.1	0.2	0.3	0.3	0.5	0.6

*where the total area for Acre is about 62279km^2, the total area for Madre de Dios is about 49348 km^2, the total area for Pando is about 54287 km^2.

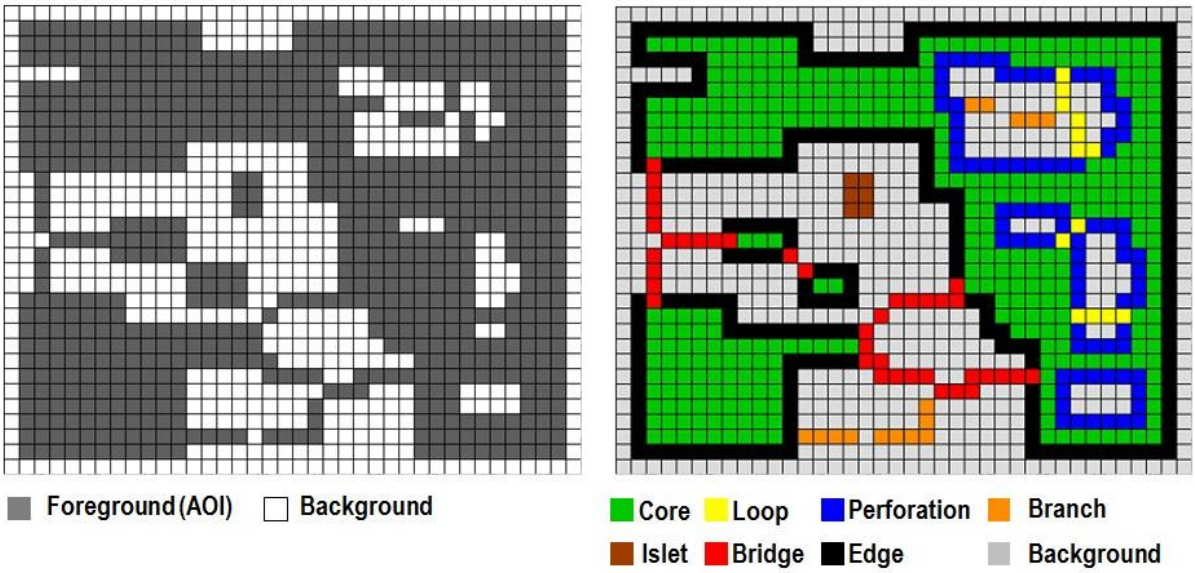

Figure 5-1. Example of morphological segmentation of binary patterns. Left: input binary pattern (foreground – forest, background – developed area). Right: Resulting segmentation. Based on the spatial arrangement, forest pixels are reclassified into seven classes: core, islet, loop, bridge, perforation, edge, and branch, while developed areas pixels were simply treated as a single class. Note: Adapted from "Morphological segmentation of binary patterns," by Soille, P. and Vogt, P., 2009, Pattern Recogn. Lett. 30 (4), 456-459. Adapted with permission.

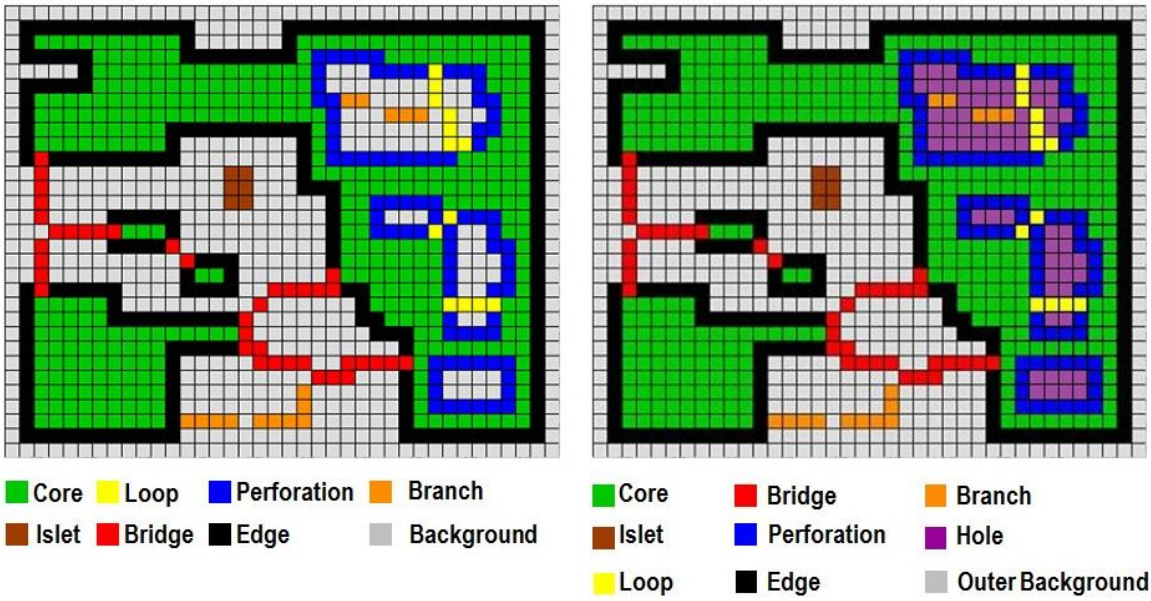

Figure 5-2. Example of MSPA patterns. Left: MSPA segmentation result. Right: Proposed hole and outer background additions. Note: Adapted from "Morphological segmentation of binary patterns," by Soille, P. and Vogt, P., 2009, Pattern Recogn. Lett. 30 (4), 456-459. Adapted with permission.

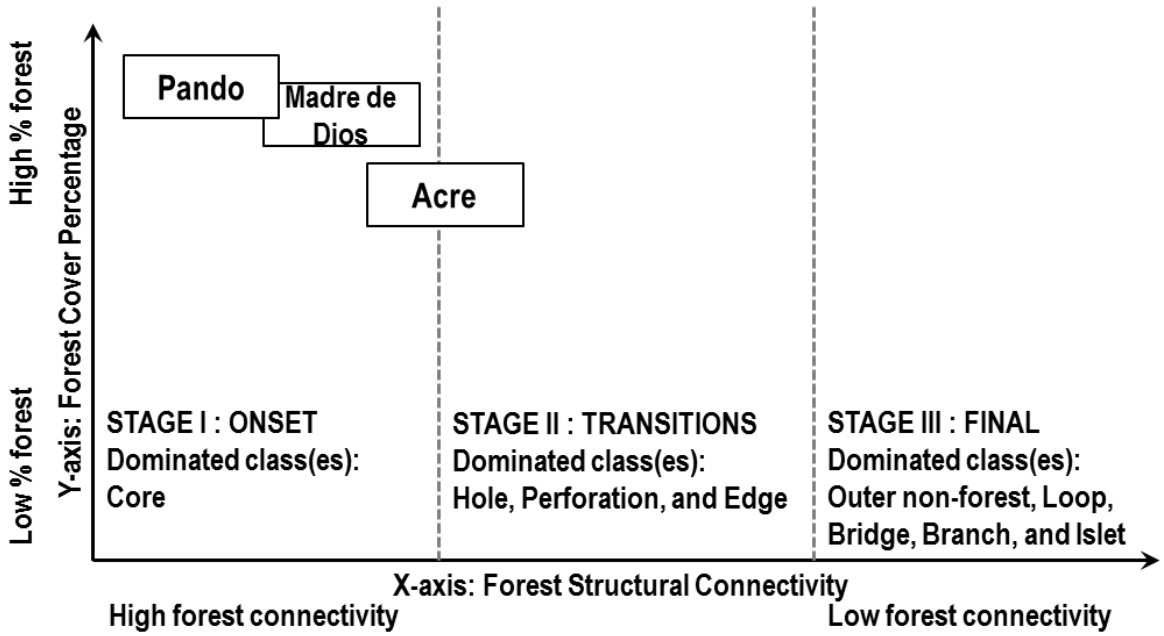

Figure 5-3. Proposed conceptual model of forest structural connectivity and composition/amount versus mathematical morphological indices, as indicators of forest landscape change and trajectory. Our three study regions are located within this model for Stages I (Pando and Madre de Dios) and Stage II for Acre.

CHAPTER 6
MAPPING FRACTALITY DURING THE PROCESS OF DEFORESTATION IN AN AMAZON TRI-NATIONAL FRONTIER[†]

Background

Human society is undergoing a rapid technique revolution that will eventually enter a post-industrial era and this drastic transition has already produced threatening changes in every segment of social-ecological systems (Gunderson and Holling, 2002) where abrupt global environmental changes can no longer be excluded (Scheffer et al., 2001, 2009). In coping with such multi-face problems, Rockström et al. (2009a, b) proposed an earth system framework 'planetary boundaries' with an attempt to numerically estimate a safe operating space for humanity, in which they argued that people must stay within the boundaries where transgressing the boundaries could be dangerous and catastrophic. According to the 'planetary boundaries' research, change in land use across the globe may soon be approaching the upper limit of safe operating boundaries, so that much attention should be given in this section to avoid potential catastrophic shift (Carpenter et al., 2009, Scheffer, 2009). While land use and land cover changes are occurring globally at an increasingly unprecedented rate, impacting almost all major biomes worldwide (Lambin and Geist 2006; Gutman et al., 2004), and among these terrestrial dynamics, deforestation has received considerable attention (Nepstad et al. 1999, 2001; Laurance et al., 2001, 2002; Andersen et al., 2002; Perz et al., 2007a, b, 2008, 2012; Pfaff et al., 2007; Giles et al., 2008; Marsik et al., 2011; Southworth et al., 2011, Sun et al., 2013).

[†] "Reprinted with permission from: Sun, J., Huang, Z., Southworth, J., & Qiu, Y. (2013). Mapping fractality during the process of deforestation in an Amazon tri-national frontier. *Remote Sensing Letters*. 4:6, 589 - 598."

Deforestation has a substantially negative influence on regional and global climates (Malhi et al., 2008). For example, if the trees are removed, evapotranspiration of moisture from the canopy/vegetation back to the atmosphere will be reduced, as such more solar energy will go into the heating of the land, less into evapotranspiration, resulting in a warmer and drier environment (Wilson and Agnew, 1992). The consumption of the cleared forest produces greenhouse gases, such as carbon dioxide, methane and nitrous oxide, which play an important role in exacerbating global warming (Fearnside, 2000; Fearnside et al., 2004). Deforestation may also signal an impending biodiversity loss or even a biotic collapse (Wilcox and Murphy, 1985; Steininger et al., 2001; Nobre et al., 2009) caused by the loss of landscape connectivity, both structural and functional. This is because persistence of spatially structured species populations, meta-populations, is strongly related to landscape connectivity (Hanski and Ovaskainen, 2003), and the fragmentation breaks continuous forests into discrete patches and then increases the level of patch isolation. Lack of connectivity would reduce the capability of species movement, for example, local extinctions of certain species could not be compensated by the immigrant population from the neighbor patches, and would interfere with pollination, seed dispersal, wildlife migration and breeding (Estreguil and Mouton, 2009).

Deforestation changes forest spatial patterns which are readily apparent to the human, and such spatial patterns can be analyzed by their composition and configuration (Li and Reynolds, 1994), that is, both the elements present and how these elements are arranged (Turner et al., 2001; Wu and Hobbs, 2002). Correspondingly, deforestation refers to two processes – the first is the area of forest cleared and the

second is the patterns of this clearing. Both of these factors are important for a variety of ecological reasons – species survivability, species dispersal, species migration patterns and therefore quantitative methods are required to measure these two factors, identify their change, and incorporate them during the process of deforestation.

Since 1972, large-scale deforestation analyses become possible with the support of satellite imagery (Giles and Burgoyne, 2008). The rapid developments of geo-techniques, such as remote sensing and geographic information system (GIS), have advanced the collection and analysis of forest data. With the assistance from remote sensing and other tools, forest studies have progressed rapidly, and in turn, forested landscapes have provided many important testing grounds for the development and application of landscape ecological principles, tools and methods (Perera et al., 2007), like Island Biogeography, Meta-population. Recently, Kupfer (2012) pointed out that the widely used package FRAGSTATS released two decades ago have revolutionized the landscape analysis and entrenched landscape pattern indices in minds and statistical tool boxes of many landscape ecologists and biogeographers. On the other hand, FRAGSTATS as well as FRAGSTATS-like metrics suffer limitations, like the mismatch between patterns and processes (Cardille et al., 2005) where the true relationship between them is important for uncovering the controlling mechanisms of the systems. Hence, in order to better understand the spatial pattern during the process of deforestation, a large collection of techniques and methodologies have been developed to facilitate measurements.

Spatial enhancement performed by spatial filters provides a new way to measure the spatial patterns across multiple scales (e.g., Riitters et al., 2000, 2002; Zurlini et al.,

2006, 2007). Graph theory, as a well-suited method, can provide spatially explicit knowledge, and a growing genre of graph theory applications (Urban and Keitt, 2001; Bunn, et al., 2000; Pascual-Hortal and Saura, 2006, Minor and Urban, 2008, Minor and Gardner, 2011, Baggio et al., 2011) and software (Pajek, UCINET, LQGraph) have been published or released during the past decades. Moreover, a promising image processing technology, morphological spatial pattern analysis (MSPA) was proposed to understand the spatial patterns at a pixel level, which uses a series of image processing routines to identify different spatial patterns (Vogt et al., 2007a). MSPA has been widely employed in landscape analysis, such as corridor mapping, scale pattern measurement, landscape connectivity mapping, and green infrastructure detection (Ostapowicz et al., 2008; Riitters et al., 2007; Vogt et al., 2007b; Wickham et al., 2010). An entropy-related index “normalized spectral entropy” (Hs_n) was introduced to describe the degree of regularity (orderliness) of an ecological time-series based on its power spectrum (Zaccarelli et al., 2012, Zurlini et al., 2012). Particularly, by layer-stacking time-series NDVI (normalized difference vegetation index) images, Hs_n can be used to indicate vegetation dynamics and most importantly, display this information in maps. Sun (2013) introduced an image processing technique, the Canny-Deriche filter, to evaluate edge complexity of satellite images, which greatly promote the understainding of landscape heterogeneity across multiple spatio-temporal scales.

Based on the short review described, it is logical to conclude that there is a clear trend to move from simple (FRAGSTATS and FRAGSTATS-like) landscape metrics applied to landscapes to the use of graphs (Kupfer, 2012), which provide a more functional approach to landscape quantification and to understand landscape dynamics.

Fractal Geometry: A Promised Approach to Study Deforested Landscapes

In deforested landscapes, fragmentation can be envisaged as a surface growth process and such complex, irregular patterns cannot be described by traditional Euclidean geometry but can be measured by fractal geometry. In this research, the fixed-grid scans approach is adopted to cartographically represent the fractal structures of developed areas in the MAP region from 1986 to 2010. We will (i) produce time-series raster maps of the study area along a 25-year span which display the spatial distribution of fragmentation characterized by fractal dimension at a pixel level; and (ii) based on the created raster maps, a classification scheme will be summarized to interpret fragmentation progress from the less developed areas to the most developed ones. The products of this research will be able to support coarse scale deforestation evaluations and can be incorporated into conservation planning to serve as important references in forest management.

Fractal dimension

A fractal structure is self-similar if it looks like itself at different scales (Mandelbrot, 1977, 1983) and can be described by a scaling law. Consider an object in Euclidean dimension D, if we reduce its linear size by 1/r in each spatial direction, its measure would increase to $N = r \times D$ times the original. Next, taking the log of both sides, $\log(N) = D \times \log(r)$, and solving for D, we have equation (6 -1)

$$D = \log(N) / \log(r) \qquad (6\text{-}1)$$

where D is the dimension of the scaling law and it needs not be an integer (Theiler, 1990) especially for the real-world objects. D can be a fraction, as it is in fractal geometry, known as Hausdorff dimension that is a commonly used fractal dimension.

Fractal dimension is used to characterize the fractal objects, where fractal dimensions are between 1 and 2 for curves and between 2 and 3 for surfaces.

In a 2-dimensional plane, the fractal dimension has been characterized as a measure of the space-filling capacity of certain patterns. For instance, for an object residing in a 2-dimensional plane, the more it fills a plane, the closer it approaches the dimension 2. In addition, values of less than 1 are also possible, indicating very small and isolated objects occurring across the 2-dimensional plane. As the value increases the size of the objects increases and also their consolidation with each other such that as we approach the maximum values (2 would be an absolute maximum, but generally the range 1.8 - 2 approaches this maximum) we are often looking at very consolidated, compact regions (Theiler, 1990).

Box-counting method

Box counting is a widely used method to calculate fractal dimension. As a sampling process, the basic procedure is to recursively superimpose a series of regular grids of decreasing box sizes over a target object (developed areas here), and then record the counting for each successive box, where the counting equals to how many of the boxes are occupied by the target object. Theoretically, the logarithm of N, the number of occupied cells, versus the logarithm of 1/r, where r is the size of one cell (the length of one side), gives a line whose gradient corresponds to the box dimension D. For developed areas in the interpreted images, the fractal dimensions can be used to indicate their spatial-filling capacity, that is, to what extent developed areas can occupy the entire box area. The more clearings and conversion into developed areas, the closer the fractal dimension approaches the value 2 within each grid. In practice, a series of grids is recursively generated within an AOI box which has been laid over developed

areas. The size of the grid is taken as $r = 1/(2^n)$, where n is an integer ranging from 1 to m, and we set the iteration number as m = 7 based on the finest grid, $r = 1/(2^7)$ or 1/128 of the original box size, which is fine enough to approximate developed areas in the MAP region and thus able to yield stable results. In this research, we superimposed a matrix of 9 km × 9 km non-overlapping fixed grids on the developed areas in the MAP region for the year 1986, 1991, 1996, 2000, 2005, and 2010 and then calculated the fractal dimension by the box-counting method within each grid. 9 km is an arbitrary choice, but is based on careful considerations: the forest fragmentation maps are interpreted from Landsat images with the smallest pixel being 30 m and thus after recursively dividing the maps ($2^7 = 128$) to calculate the fractal dimension, the finest grid should be larger than the map resolution 30 m. Moreover, for calculation convenience, 9 km could be divided exactly by 30 m without remainders. Finally, the estimation of the linear regression, $\ln N = D \times \ln r + c$, is implemented in R (the R foundation for Statistical Computing), where c is a constant, both the coefficient D and the coefficient of determination R^2 are recorded to measure the fractal dimension and to describe how well the regression fits.

Cartographic Representation of Fractal Structure across the MAP Region

The maps of fractal dimension in Figure 6-1 support that the landscape is heterogeneous, as the range of fractal dimensions are not evenly distributed, and such a property thus indicates that a single fractal dimension calculated by a single large AOI (box) would blur much of this information. The resultant figure reveals that the cleared areas in the MAP region have become increasingly compact from 1986 to 2010, where the low fractal dimensions typically represent little to no forest clearing, and higher fractal dimensions are associated with more highly developed areas.

The fractal dimension 1.7 is a worldwide average of urban built-up areas (Batty and Longley 1994, Alfasi and Portugali, 2004), and based on the findings of Chen (2000) who employed a wave-spectrum analysis to estimate the fractal dimension and concluded that the closer the fractal dimension is to 1.7, the better estimated the results will be, so that the boxes with 1.7 or higher fractal dimensions are highlighted to indicate these regions of highly developed areas. The total highlighted areas increases from 8 grids (each grid is 81 km^2) in 1986 to 217 grids in 2010, and this process in terms of timeline was seen first in Rio Branco (capital of Acre) in 1986, and then Cobija (capital of Pando) in 1991, and then finally in Puerto Maldonado (capital of Madre de Dios) by 1996. By 2010, there are 193 highlighted grids in Acre, 15 in Madre de Dios, and 9 in Pando, which corresponds to the different levels and rates of development and infrastructure across the three sides of the border. In addition, to estimate the quality of the fractal analysis, goodness-of-fit (R^2) is computed and summarized in Figure 6-1, where the large R^2 distributions can be seen to support the robust regression results and fractal structures.

Previous research across this landscape (Southworth et al. 2011; Perz et al. 2012) has highlighted the increasing patterns of deforestation, which do vary across the national frontiers, but all of which are dominated by forest clearing leading to forest fragmentation, and only extremely limited amounts of forest regrowth. While these levels of deforestation are still quite low (compared to global rates, over 80% forest cover is considered a high amount of forest cover) it is still of great importance to better understand the spatial patterns of these changes, especially across such dynamic landscapes. Intending to understand forest fragmentation as a development process,

we propose a framework to classify the continuous fractal dimensions into four categories and this is summarized in Figure 6-2.

The classification scheme interprets fractal dimension into four levels based on the amounts, patterns and arrangements of the forest clearings or developed areas. Level I ($0 \leq D < 1.0$) presents very minimal developed areas and they are quite dispersed, not revealing any substantial patterns. Once we move beyond the level of $D \geq 1.0$, developed areas start to exhibit linear patterns, in part as within this region many of the clearings and associated development occurs along roads. Level II ($1 \leq D < 1.7$) we found that developed areas change from quite disperse to compact as these regions of clearings start to agglomerate. Level III ($1.7 \leq D < 1.8$) is where we really begin to observe clump consolidation as a common feature, and globally, this level is considered to equate to urbanized regions i.e. highly developed. For this research we also chose to add a Level IV, in part as the data highlighted differences here which we felt were worthy of a separate class, here level IV ($1.8 \leq D \leq 2.0$) appears to be made up of mostly consolidated regions of developed areas. Using this classification scheme, developed from global studies, facilitates the comparison of this region both internally and also with other global models of deforestation development.

Summary

Amazon deforestation has often been seen as the most prominent galvanizing image of environmental change during the last several decades, and thus it is not easy to position this research within the literature. Focusing on the MAP region, however, Peralta and Mather (2000) interpreted multi-temporal forest/non-forest maps in Acre and then calculated landscape metrics to characterize the impacts of social and economic processes on the development of the forest landscape. Millington et al. (2003)

measured different spatial patterns of forest fragmentation in Bolivia, and explained these patterns with reference to specific government policies. Moreover, their results also indicated that most landscape metrics were highly dependent on pixel size. Similarly, Bradley and Millington (2006) discussed the mismatch between the spatial scale of biomass burning and the satellite products in Bolivia and Peru and found that an entire component of the fire region in the study region is omitted, despite its importance in the farming systems.

The MAP region provides an ideal case for study because it contains high ecological and cultural diversity, while entering a period of rapid change triggered by the paving of the Inter-Oceanic highway. The road paving in Brazil has been completed to the Peruvian and Bolivian borders, which has greatly stimulated local economic development. New legislation for logging concessions in Peru and Bolivia has fostered increased logging, and consequently, land values have risen (Wadt et al., 2005). However, this tri-national frontier is still largely forested and has been designated a global biodiversity hotspot (Myers et al., 2000).

In the face of increasing environmental changes, we emphasize that landscape management should focus on the local solutions instead of a global optimization and therefore foster local conservation management as a way to withstand disturbances and to maintain system stability. Currently, the expansion of clearings in the MAP region is accompanied by some early urbanized areas, however, the binary forest/non-forest maps produced cannot reflect this change. The theoretically supported fractal dimension of 1.7 (Chen, 2010) serves as a useful value to slice the fragmentation stages, but it just describes the spatial patterns of developed areas and cannot indicate

urbanization here, as the input data is not city boundaries but rather is cleared/developed areas which include human settlements, roads, agriculture, bare soil, etc. To better understand the interaction between deforestation and urbanization (DeFries et al., 2010) in this landscape, independent statistical data, such as economic, social, or demographic data, should be incorporated into any future analysis. The cartographic representations of fractal dimension and classification schemes presented are simple enough to be applied in any fragmented area, which provides an alternative perspective to the spatial methods devised to date.

The proposed fixed-grid scans methodology pixelizes the entire landscape into a matrix of grids and calculates the fractal dimension within each grid, which expands the application of this approach in coarse scale studies and prompts the understanding of fragmentation by fractals across this heterogeneous landscape. Since forest clearings are not homogeneous across the MAP region, the classification scheme proposed further partitions the continuous fractal dimensions into four categories, that is, from no or very small developed areas occurring in a highly forested landscape (level I), to where developed areas have initiated but are still quite patchy (level II), then to a process of increasing forest clearings, where fragmentation converts over to clumping of cleared patches (level III) and then finally into clearing consolidation (level IV). As such, we define the fragmentation/development level according to fractal dimension and this could be utilized to provide useful context to conservation and management programs. Development of these ideas and models could be further developed and tested, as just one potential indicator of significant landscape changes. This could then provide important information to conservation or management groups of the types of changes

already in progress, their rates of change and ideally an idea of future trajectories (assuming no change in current patterns/trends). In addition, different levels of deforestation identified by the model require conservation management to be flexible and adaptive. Such planning tools seem imperative in these fragile landscapes and biodiversity hotspots.

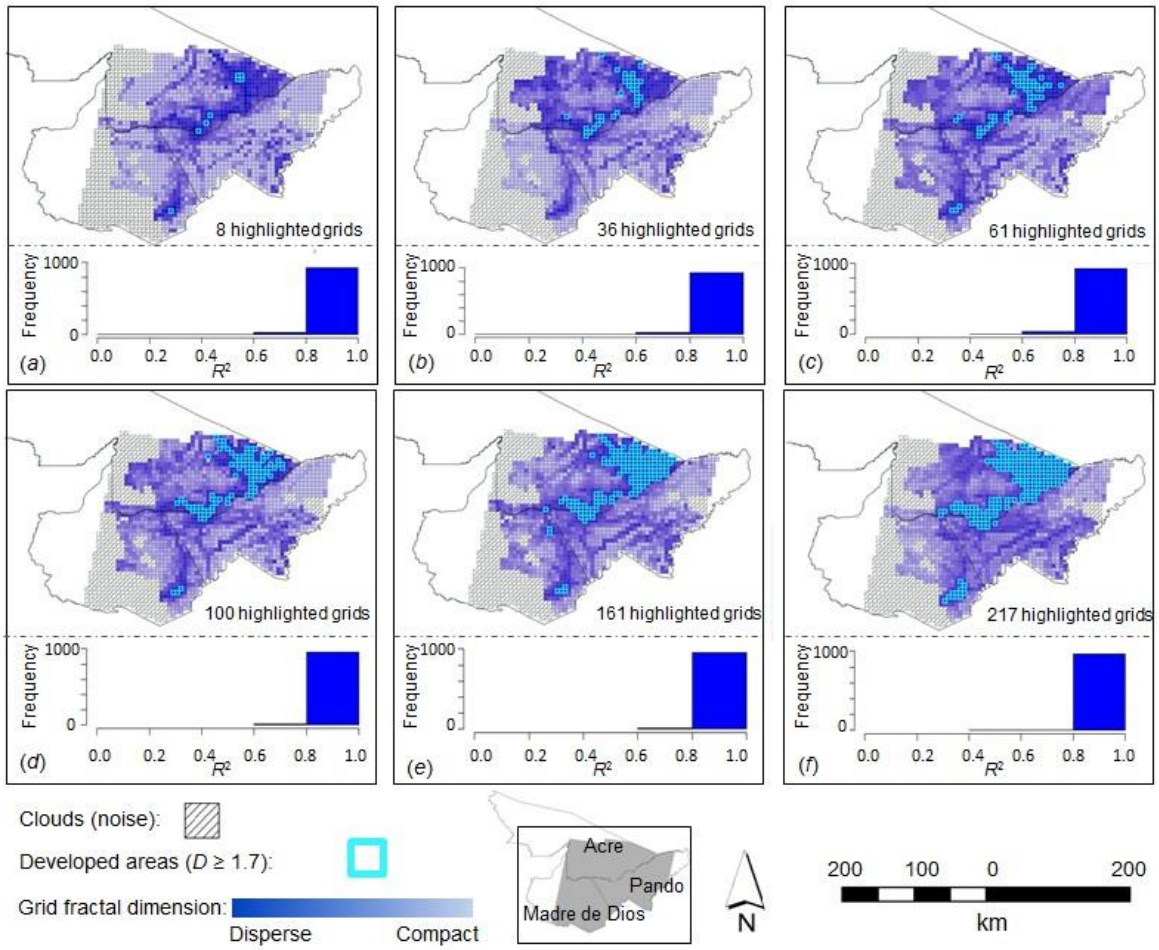

Figure 6-1. Spatio-temporal fractal analysis of the cleared areas in the MAP region for the years (A) 1986. B) 1991.C) 1996. D) 2000. E) 2005. F) 2010. The boxes with fractal dimensions equal or above 1.7 are highlighted in light blue to indicate intensively developed areas and their evolving processes. To maintain the data quality and result accuracy, some grids were removed due to cloud influence. In addition, the distributions of R^2 of the linear regression in the box-counting calculation within each grid are displayed, where a large R^2 suggests a robust fractal property.

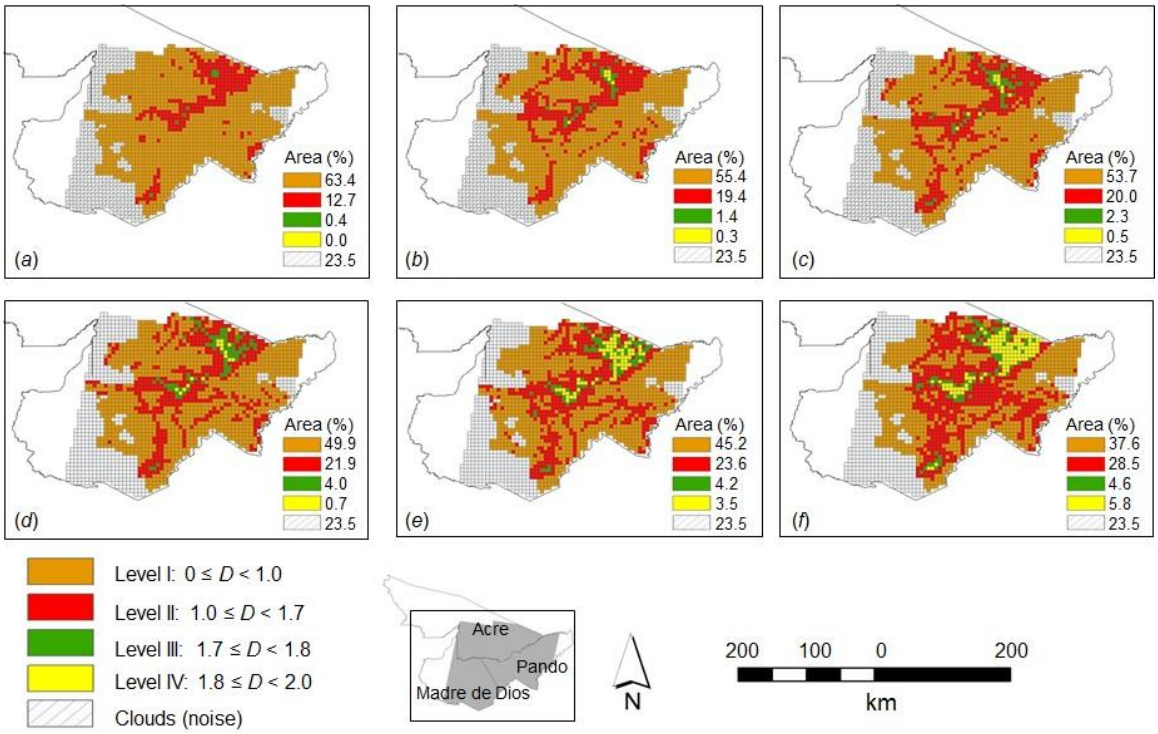

Figure 6-2. The evolving pattern of development as indicated by density-sliced fractal dimension in the MAP region for the years A) 1986. B) 1991.C) 1996. D) 2000. E) 2005. F) 2010, where the level I decreases from 63.4% in 1986 to 37.6% in 2010, and level II, III, and IV increase from 12.7%, 0.4%, 0.0% in 1986 to 28.5%, 4. 6%, and 5.8% in 2010 and 1% represents 1655.9 km2 in the entire MAP region.

CHAPTER 7
CONCLUSIONS

Operating within the resilience paradigm, this study assesses and reports spatial pattern changes during the process of deforestation in the Amazonian rainforest, aimed to guide forest evaluation and conservation programs. Since forest patterns are readily detectable from remotely sensed images, and are thus amenable to study at broad scales, they have provided many opportunities for studying the complex relationship between patterns and processes in many areas. After a brief review of knowledge on resilience theory to address spatial pattern process likely to have ecological effects during the process of deforestation, three measures were illustrated including their roles as proxies to indirectly assess forest landscape resilience.

What Can We Tell the People from This Research?

Changes in forest spatial patterns are readily apparent to the human eye, which alters the composition and configuration of forest landscapes and this, in turn, influences many species and ecosystem processes. Therefore, the importance of landscape-level considerations in the management and conservation of forested landscapes has become increasingly important, and a variety of stakeholders are involved. The discipline of landscape ecology has developed concepts and methodologies that can be directly applied in forested landscapes, but to be most useful, these need to be more widely available. Included are principles and theory that relate spatial patterns at multiple spatial and temporal scales to ecological and anthropogenic drivers; methods for quantifying and evaluating spatial patterns in both discrete and continuous variables; and models for projecting the consequences of alternative scenarios.

Three ‘maps’ derived from this research provide relatively complete information on the spatial pattern changes and spatial distribution of the forest landscape, including forest areas and developed areas, which is not only useful for the MAP region and the Amazon, but also applicable for deforested landscapes around the world. However, it can be observed that the difficulties faced by forestry professionals in their attempts to apply these landscape ecological concepts, models real world problems. It is important to develop and synthesize landscape ecological knowledge and transfer it to users to ensure appropriate application:

- The product of Chapter 4, measured by a combination of moving window analysis and ISO unsupervised classification, can be used to evaluate the influences of forest clearings expressed by land cover changes across multiple spatial scales. It notifies people that forest clearings are not limited by the physical boundaries but decreasingly spread over a larger scope, and the mappings of these impacted areas and their associated impact levels help make forest conservation programs better able to apply these data, both locally and effectively.

- In Chapter 5, measures of forest spatial patterns at a pixel level were based on the application of MSPA methods, and the existing MSPA method was improved by adding two classes which focus on the spatial properties of developed areas. This product strengthens the understanding of forest configuration by emphasizing spatial properties. In addition, the proposed conceptual model can be used to indicate the dynamics of landscape structural connectivity based on the MSPA results.

- Besides the two additional tools of MSPA proposed in Chapter 5, the spatial patterns of developed areas were further characterized by fractal geometry in Chapter 6. The fractal dimensions were calculated across the entire landscape by adopting a fixed-grid scans approach and summarized by a classification scheme to partition deforestation patterns/amounts into different ‘levels’. Such cartographic representation of fractal structures provides flexible and adaptive information about forest clearings to forest managers and to more regional land use planning.

By providing three measures and illustrating them with the real world data in the MAP region, this research makes a significant contribution to the applications of

landscape ecology in deforested landscapes as well as promoting the understanding of forest resilience with particular attention being paid to the spatial patterns. These results should encourage the application of the methods and measures for large areas assessment and reporting. For biodiversity purposes, it will be essential to further complement local change in forest pattern with field-based assessments of forest quality. This was not feasible to undertake in this study.

Knowledge transfer is necessary but not sufficient for applications of landscape ecology to be successful. The chapters within demonstrate the potential possibility to translate the science of landscape ecology into practice. This research should be of broad interest to all those interested in understanding and managing forested landscapes, and in particular to current and future researchers committed to making the knowledge they develop available to a wider audience of users.

Resilience Prompts the Understanding of Unexpected Shifts

Human society is undergoing a rapid technical revolution that will eventually enter a post-industrial global information age and this drastic transition has already produced threatening changes in the every segment of social-ecological systems where abrupt global environmental changes can no longer be excluded. In coping with such multi-faceted problems, Rockström et al. (2009) proposed an earth system framework of ‘planetary boundaries’ to numerically estimate a safe operating space for humanity with a holistic perspective (Naveh, 2000, 2001), and they argued that people must stay within the boundaries, as transgressing the boundaries could be dangerous and catastrophic. Therefore, plans to understand complex problems of social-ecological systems should always consider system boundaries (Carpenter et al., 2009) and regime shift prevention.

Regime shifts are large, sudden changes in ecosystems that last for substantial periods of time (Scheffer et al., 2001, Carpenter, 2003, Biggs et al., 2009), caused by the combination of the internal resilience of the system and the magnitudes of external forces (Folke et al., 2004). Regime shifts in ecosystems can cause large losses of ecological and economic resources, and restoring such systems to a 'desired state' may require drastic and expensive intervention, which is known as system hysteresis. From a practical point of view, hysteresis is important, as it implies this kind of catastrophic transition is not so easy to reverse (Scheffer et al., 2001, 2009; Scheffer, 2009) and therefore, managers should be especially vigilant about trying to prevent or avoid regime shifts. Based on experiments, historical data, and theoretical models, in an attempt to signal the surprising in the systems, some early warning indicators have been obtained to announce the coming regime shifts in advance (Carpenter et al., 2011), such as rising variance (Carpenter and Brock, 2006), rising autocorrelation (Dakos et al. 2008), and extreme changes in skewness and variance (Guttal and Jayaprakash, 2008, 2009). But, when these indicators become statistically significant and are able to be detected and verified, systems have already lost a certain amount of resilience. For those systems with severely degraded resilience, they may not be able to withstand external disturbances (could be a slight disturbance) even after human interventions (such as disturbance-prevention managements), and could therefore be tipped into an alternative regime, often less desirable. Several studies have shown that conventional approaches to resource and ecosystem management are not always working, like disturbance control strategy (Scheffer et al., 2001), and may indeed make some problems even worse, but they may be solved if we embrace 'resilience thinking'

(Walker and Salt, 2006) in our creation of tools and techniques developed to monitor and warn of just such potentially catastrophic and irreversible regime shifts or tipping points.

Embracing Resilience Thinking

The concept of resilience has evolved considerably since Holling's (1973) seminal paper, and while Walker et al. (2004) pointed out that different interpretations of what is meant by resilience can cause confusion, they then attempted to clarify this concept in terms of the attributes that govern the system's dynamics. However, a lot of research uses resilience now, and it seems to mean different things, depending on the research context. Such 'vagueness' totally disagrees with the traditional philosophy of science (e.g., Vienna Circle, Karl Popper, Frankfurt school) that declares conceptual clarity as essential for scientific research. After comparing with the traditional school of thought (who 'only do precision'), Strunz (2012) investigated and concluded that the 'vagueness' of resilience is an asset for creative communication and pragmatic problem-solving. Strunz (2012) proposed that resilience thinking can be framed to take advantage of its vagueness without forgoing precision where it matters, which greatly enriches and solidifies the paradigm of resilience. To appreciate the complex ecosystem problems like the potential regime shift described, the vagueness of resilience thinking may provide a road map that is able to promote interdisciplinary integration which addresses the dynamics and development of complex social-ecological systems and solves the problems from a holistic perspective (Naveh, 1998, 2000; Folke et al., 2010; Walker and Salt, 2006).

In the face of increasing environmental changes, it is important to emphasize that ecosystem management should concentrate on not only the disturbance preventions

but also the gradual changes that affect system resilience and thus building resilience to withstand disturbances and to maintain a desired system state is likely to be an alternative pragmatic and effective way to manage ecosystems. Then, ‘basins of attraction’ in a stability landscape becomes a useful metaphor to visually demonstrate system dynamics and the way to enhance resilience, where the system is a ball in the basin, and the resilience is the maximum perturbation that can be taken without tipping the ball to an alternative basin (Walker et al., 2004; Scheffer et al., 2001). To increase system resilience equals making the desirable basin wider and deeper, consequently, reducing the probability of the ball being tipped outside the basin.

As has become clear, the dissertation is critical towards using the resilience framework in explaining deforestation, and it has answered key questions of our subject. Therefore, it is useful to take up resilience theory in geography study, which has added numerous values to geography, both physical geography and human geography.

LIST OF REFERENCES

Achard, F., Eva, H.D., Stibig, H.-J., Mayaux, P., Gallego, J., Richards, T., and Malingreau, J.-P. (2002), “Determination of deforestation rates of the world's humid tropical forests,” *Science*, 297(5583), 999-1002.

Alfasi, N., and Portugali, J. (2004), “Planning just-in-time versus planning just-in-case,” *Cities*, 21, 29–39.

Andersen, L.E., Granger, C.W.J., Reis, E.J., Weinhold, D., and Wunder, S. (2002), *The dynamics of deforestation and economic development in the Brazilian Amazon*. Cambridge: Cambridge University Press.

Baggio, J., Salau, K., Janssen, M., Schoon, M., and Bodin, O. (2011), “Landscape connectivity and predatorprey population dynamics,” *Landscape Ecology*, 26, 33–45.

Batty, M., and Longley, P. (1994), *Fractal Cities*. Waltham: Academic Press.

Biggs, R., Carpenter, S.R., and Brock, W.A. (2009), “Turning back from the brink: detecting an impending regime shift in time to avert it,” *Proceedings of the National Academy of Sciences*, 106, 826-831.

Bogaert, J., Ceulemans, R., and Eysenrode, S.D. (2004), “Decision tree algorithm for detection of spatial processes in landscape transformation,” *Environmental management*, 33(1), 62-73.

Bradley, A.V., and Millington, A.C. (2006), “Spatial and temporal scale issues in determining biomass burning regimes in Bolivia and Peru,” *International Journal of Remote Sensing*, 27, 2221–2253.

Brook, B.W., Sodhi, N.S., and Ng, P.K.L. (2003), “Catastrophic extinctions follow deforestation in Singapore,” *Nature*, 424(6947), 420-426.

Brown, B.E., Clarke, K.R., and Warwick, R.M. (2002), “Serial patterns of biodiversity change in corals across shallow reef flats in Ko Phuket, Thailand, due to the effects of local (sedimentation) and regional (climatic) perturbations?” *Marine Biology*, 141, 24-29.

Bunn, A.G., Urban, D.L., and Keitt, T.H. (2000), “Landscape connectivity: A conservation application of graph theory,” *Journal of Environmental Management*, 59(4), 265-278.

Cardille, J., Turner, M., Clayton, M., Gergel, S., and Price, S. (2005), “METALAND: Characterizing spatial patterns and statistical context of landscape metrics,” *Bioscience*, 55, 983–988.

Carpenter, S.R. (2003), “The need for fast and frugal models,” in *Models in Ecosystem Science*, eds. C. Canham, J. Cole and W. Lauenroth, Princeton: Princeton University Press.

Carpenter, S.R., and Brock, W.A. (2006), “Rising variance: a leading indicator of ecological transition,” *Ecology Letters*, 9, 311–318. http://dx.doi.org/10.1111/j.1461-0248.2005.00877.x

Carpenter, S.R., Cole, J.J., Pace, M.L., Batt, R.D., Brock, W.A., Cline, T., Coloso, J., Hodgson, J.R., Kitchell, J.F., Seekell, D.A., Smith, L., and Weidel, B. (2011), “Early warnings of regime shifts: A whole-ecosystem experiment,” *Science*, 332, 1079-1082.

Carpenter, S., Mooney, H., Agard, J., Capistrano, D., DeFries, R., Diaz, S., Dietz, T., Duriappah, A., Oteng-Yeboah, A., Pereira, H. M., Perrings, C., Reid, W. V., Sarukhan, J., Scholes, R. J., and Whyte, A. (2009), “Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment,” *Proceedings of the National Academy of Sciences*, 106, 1305-1312.

Carpenter, S.R., Westley, F., and Turner, M.G. (2005), “Surrogates for resilience of social–ecological systems,” *Ecosystems*, 8, 941–944.

Chen, Y. (2010), “Exploring the fractal parameters of urban growth and form with wave-spectrum analysis,” *Discrete Dynamics in Nature Society*, Article ID 974917, 20 page. doi: 10.1155/2010/974917.

Cumming, G.S., Barnes, G., Perz, S., Schmink, M., Sieving, K., Southworth, J., Binford, M., Holt, R., Stickler, C., and Van Holt, T. (2005), “An exploratory framework for the empirical measurement of resilience,” *Ecosystems*, 8, 975-987.

Cumming, G.S., Barnes, G., and Southworth, J. (2008), “Environmental asymmetries,” in *Complexity Theory for a Sustainable Future (Complexity in Ecological Systems)*, eds. J. Norberg, and G.S.Cumming, New York: Columbia University Press.

Cumming, G.S., Southworth, J., Rondon, X.J. and Marsik, M. (2012), “Spatial complexity in fragmenting Amazonian rainforests: Do feedbacks from edge effects push forests towards an ecological threshold?” *Ecological Complexity*, 11, 67-74.

Dakos, V., Scheffer, M., van Nes, E. H., Brovkin, V., Petoukhov, V., and Held, H. (2008), “Slowing down as an early warning signal for abrupt climate change,” *Proceedings of the National Academy of Sciences*, 105(38), 14308-14312.

DeFries, R.S., Houghton, R. A., Hansen, M. C., Field, C. B., Skole, D., and Townshend, J. (2002), “Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 1990s,” *Proceedings of the National Academy of Sciences*, 99(22), 14256-14261.

DeFries, R.S., Rudel, T., Uriarte, M., and Hansen, M. (2010), “Deforestation driven by urban population growth and agricultural trade in the twenty-first century,” *Nature Geoscience*, 3*,* 178-181.

Demurger, S. (2001), “Infrastructure and economic growth: An explanation for regional disparities in China?” *Journal of Comparative Economics*, 29, 95–117.

Dunning, J.B., Danielson, B.J., and Pulliam, H.R. (1992), “Ecological processes that affect populations in complex landscapes,” *Oikos*, 65, 169-175.

Dwivedi, R.S., Sreenivas, K., and Ramana, K.V. (2005), “Cover: Land-use/land-cover change analysis in part of Ethiopia using Landsat thematic mapper data,” *International Journal of Remote Sensing*, 26, 1285-1287.

Estrada, E., and Bodin, Ã. (2008), “Using network centrality measures to manage landscape connectivity,” *Ecological Applications*, 18(7), 1810-1825.

Estreguil, C., and Mouton, C. (2009), “Measuring and reporting on forest landscape pattern, fragmentation and connectivity in Europe: methods and indicators,” Joint Research Centre, European Communities*, Scientific and Technical Research series* – EUR 23841EN; ISSN 1018-5593, pp. 69.

Fan, S., Zhang, L., and Zhang, X. (2004), “Reform, investment, and poverty in China,” *Economic Development and Cultural Change*, 52, 395–421.

Fearnside, P.M. (1996), “Amazonian deforestation and global warming: carbon stocks in vegetation replacing Brazil's Amazon forest,” *Forest Ecology and Management*, 80(1-3), 21-34.

Fearnside, P.M. (2000), “Global warming and tropical land-use change: Greenhouse gas emissions from biomass burning, decomposition and soils in forest conversion, shifting cultivation and secondary vegetation,” *Climate Change*, 46, 115–145.

Fearnside, P.M. (2004), “Greenhouse gas emissions from hydroelectric dams: controversies provide a springboard for rethinking a supposedly “clean” energy source,” *Climatic Change*, 66(2–1), 1–8. http://dx.doi.org/10.1023/B:CLIM.0000043174.02841.23.

Fearnside, P.M. (2005), “Deforestation in Brazilian Amazonia: History, Rates, and Consequences,” *Conservation Biology*, 19(3), 680-688.

Folke, C., Carpenter, S.R., Walker, B., Scheffer, M., Chapin, T., and Rockström, J. (2010), “Resilience thinking: integrating resilience, adaptability and transformability,” *Ecology and Society*, 15(4): 20. [online] URL: http://www.ecologyandsociety.org/vol15/iss4/art20/

Folke, C., Carpenter, S.R., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., and Holling, C. S. (2004), "Regime shifts, resilience, and biodiversity in ecosystem management," *Annual Review of Ecology, Evolution, and Systematics*, 35, 557–581

Foody, G.M., and Curran, P.J. (1994), "Estimation of tropical forest extent and regenerative stage using remotely sensed data," *Journal of Biogeography*, 21, (3), 223-244.

Forman, R.T.T. (1995), "Some general principles of landscape and regional ecology," *Landscape Ecology*, 10, 133-142.

Foster, D.R., Motzkin, G., and Slater, B. (1998), "Land-use history as long-term broad-scale disturbance: regional forest dynamics in central New England," *Ecosystems*, 1, 96–119.

Gardner, R. (1999), "RULE: A program for the generation of random maps and the analysis of spatial patterns," in: *Landscape ecological analysis: issues and applications*, eds. J.M. Klopatek, R.H. Gardner, New York: Springer-Verlag, pp 280–303.

Gibson, J., and Rozelle, S. (2003), "Poverty and access to roads in Papua New Guinea," *Economic Development and Cultural Change*, 52, 159–185.

Giles, P.T., and Burgoyne, J.M. (2008), "Skole, D.L. and Tucker, C.J. 1993: Tropical deforestation and habitat fragmentation in the Amazon: satellite data from 1978 to 1988. Science 260, 1905—1910," *Progress in Physical Geography*, 32(5), 575-580.

Gragson, T. (1998), "Potential versus actual vegetation: Human behavior in a landscape medium," in *Advances in Historical Ecology*, eds. W. Ballee, New York: Columbia University Press, pp. 213-231.

Gunasekera, K., Anderson, W., and Lakshmanan, T.R. (2008), "Highway-induced development: Evidence from Sri Lanka," *World Development*, 36, 2371–2389.

Gunderson, L. (2010), "Ecological and human community resilience in response to natural disasters," *Ecology and Society*, 15(2): 18. [online] URL: http://www.ecologyandsociety.org/vol15/iss2/art18/.

Gunderson, L, and Holling. C.S. (2002), *Panarchy: Understanding Transformations in Human and Natural Systems*, 1st Edition. Washington D.C.: Island Press.

Gunderson, L., and Pritchard, L. (2002), *Resilience and the behavior of large-scale systems*. Washington D.C.: Island Press.

Gutman, G., Janetos, A., Justice, C., Moran, E., Mustard, J., Rindfuss, R., Skole, D., and Turner, B. L. (2004), *Land change science: Observing, monitoring, and understanding trajectories of change on the Earth's surface*, the Netherlands: Kluwer Academic.

Guttal, V., and Jayaprakash, C. (2009), "Spatial variance and spatial skewness: leading indicators of regime shifts in spatial ecological systems," *Theoretical Ecology*, *2*(1), 3-12.

Guttal, V., and Jayaprakash, C. (2008), "Changing skewness: an early warning signal of regime shifts in ecosystems," *Ecology letters*, 11(5), 450-460.

Hansen, A.J., and Rotella, J.J. (2002), "Biophysical factors, land use, and species viability in and around nature reserves," *Conservation Biology*, 16, 1-12.

Hanski, I. (1999), *Metapopulation ecology*, Oxford, Oxford University Press.

Hanski, I., and Ovaskainen, O. (2003), "Metapopulation theory for fragmented landscapes,". *Theoretical Population Biology*, 64, 119-127.

Holling. C.S. (1973), "Resilience and stability of ecological systems," *Annual Review of Ecology and Systematics*, 4, 1-23.

Holling, C.S. (1996), "Engineering resilience versus ecological resilience," in Engineering within ecological constraints, eds. P. Schulze, Washington, D.C.: National Academy Press, pp. 31-44.

Hughes, T.P. (1994), "Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef," *Science*, 265, 1547-1551.

Killeen, T.J., and Solórzano, L.A. (2008), "Conservation strategies to mitigate impacts from climate change in Amazonia," *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, 363(1498), 1881-1888.

Kupfer, J.A. (2012), "Theory in landscape ecology and its relevance to biogeography," in *The SAGE Handbook of Biogeography*, eds. A.Millington, M. Blumler, and U. Schickhoff, Newbury Park: Sage Press, pp. 57–74.

Lambin, E.F., and Geist, H.J. (2006), *Land-use and land cover change: Local processes and global impacts*. Berlin: Springer.

Laurance, W.F., Albernaz, A.K., and Da Costa, C. (2001), "Is deforestation accelerating in the Brazilian Amazon?" *Environmental Conservation*, 28(04), 305-311.

Laurance, W.F., Albernaz, A.K., Fearnside, P. M., Vasconcelos H. L., and Ferreira l. V. (2004), "Letters to the Editor: Deforestation in Amazonia," *Science*, 304, 21.

Laurance, W.F., Albernaz, A.K., Schroth, G., Fearnside, P. M., Bergen, S., Venticinque, E. M., and Da Costa, C. (2002), "Predictors of deforestation in the Brazilian Amazon," *Journal of Biogeography*, 29(5-6), 737-748.

Levin, S.A. (1992), "Orchestrating environmental research assessment," *Ecological Applications, 2*, 103-106.

Lewontin, R.C. (1969), *The Meaning of Stability, Diversity and Stability in Ecological Systems*. New York: Brookhaven National Laboratory.

Li, H., and Reynolds, J.F. (1994), "A simulation experiment to quantify spatial heterogeneity in categorical maps," *Ecology*, 75, 2446–2455.

Malhi, Y., Roberts, J.T., Betts, R.A., Killeen, T. J., Li, W., and Nobre, C.A. (2008), "Climate change, deforestation, and the fate of the Amazon," *Science*, 319, 169–172.

Mandelbrot, B.B. (1977), *Fractals, Form, Chance and Dimension*, San Francisco: W.H. Freeman and Company.

Mandelbrot, B.B. (1983), *The Fractal Geometry of Nature*, New York: Freeman and Company.

Marsik, M., Stevens, F.R., and Southworth, J. (2011), "Amazon deforestation: Rates and patterns of land cover change and fragmentation in Pando, northern Bolivia, 1986 to 2005," *Progress in Physical Geography*, 35(3), 353-374.

Mas, J.-F., Velázquez, A., Díaz-Gallegos, J.R., Mayorga-Saucedo, R., Alcántara, C., Bocco, G., Castro, R., Fernández, T., and Pérez-Vega, A. (2004), "Assessing land use/cover changes: a nationwide multidate spatial database for Mexico," *International Journal of Applied Earth Observation and Geoinformation*, 5, 249-261.

May, R.M. (1977), "Thresholds and breakpoints in ecosystems with a multiplicity of states," *Nature*, 267(5628): 471–477. http://dx.doi.org/10.1038/269471a0.

McGarigal, K., Cushman, S.A, Neel, M.C., and Ene, E. (2002), "FRAGSTATS v3: Spatial Pattern Analysis Program for Categorical Maps," *Computer software program produced by the authors at the University of Massachusetts*, Amherst. Available at the following web site: http://www.umass.edu/landeco/research/fragstats/fragstats.html.

Minor, E.S., and Urban, D.L. (2008), "A Graph-Theory framework for evaluating landscape connectivity and conservation planning," *Conservation Biology*, 22(2), 297-307.

Minor, E.S., and Gardner, R. (2011), “Landscape connectivity and seed dispersal characteristics inform the best management strategy for exotic plants,” *Ecological Applications*, 21(3), 739–749.

Moran, E.F., and Ostrom, E. (2005), *Seeing the Forest and the Trees: Human-environment Interactions in Forest Ecosystems*. Cambridge: MIT Press, pp: 456.

Myers, N., Mittermeier, R. A., Mittermeier, C.G., da Fonseca, G.A.B., and Kent, J. (2000), “Biodiversity hotspots for conservation priorities,” *Nature*, 403(6772), 853-858.

Nagendra, H. (2004), “From pattern to process: landscape fragmentation and the analysis of land use/land cover change,” *Agriculture, Ecosystems and Environment*, 101(2-3), 111-115.

Nagendra, H., Southworth, J., and Tucker, C. (2003), “Accessibility as a determinant of landscape transformation in western Honduras: linking pattern and process,” *Landscape Ecology*, 18(2), 141-158.

Naveh, Z. (1998), “Ecological and cultural landscape restoration and the cultural evolution towards a Post-Industrial symbiosis between human society and nature,” *Restoration Ecology*, 6(2), 135-143.

Naveh, Z. (2000), “What is holistic landscape ecology? A conceptual introduction,” *Landscape and Urban Planning*, 50(1-3), 7-26.

Naveh, Z. (2001), “Ten major premises for a holistic conception of multifunctional landscapes,” *Landscape and Urban Planning*, 57, 26-284.

Nepstad, D., Carvalho, G., Cristina Barros, A., Alencar, A., Paulo Capobianco, J., Bishop, J., Moutinho, P., Lefebvre, P., Lopes Silva, U. and Prins, E. (2001), “Road paving, fire regime feedbacks, and the future of Amazon forests,” *Forest Ecology and Management*, 154, 395-407.

Nepstad, D., Verssimo, A., Alencar, A., Nobre, C., Lima, E., Lefebvre, P., Schlesinger, P., Potter, C., Moutinho, P., Mendoza, E., Cochrane, M., and Brooks V. (1999), “Large-scale impoverishment of Amazonian forests by logging and fire,” *Nature*, 398, 505-508.

Nobre, P., Malagutti, M., Urbano, D.F., de Almeida, R.A.F., and Giarolla, E. (2009), “Amazon deforestation and climate change in a coupled model simulation,” *Journal of Climate*, 22(21), 5686-5697.

O’Neill, R.V., De Angelis, D.L., Waide, J.B., and Allen, T.F.H. (1986), *A hierarchical concept of ecosystems*. Princeton: Princeton University Press.

Opdam, P. (2002), “Assessing the conservation potential of habitat networks,” in *Applying landscape ecology in biological conservation*, eds. K. Gutzwiller, and T.T. Forman, New York: Springer, pp. 381–404.

Ostapowicz, K., Vogt, P., Riitters, K., Kozak, J., and Estreguil, C. (2008), “Impact of scale on morphological spatial pattern of forest,” *Landscape Ecology*, 23(9), 1107-1117.

Pascual-Hortal, L., and Saura, S. (2006), “Comparison and development of new graph-based landscape connectivity indices: towards the priorization of habitat patches and corridors for conservation,” *Landscape Ecology*, 21(7), 959-967.

Peralta, P., and Mather, P. (2000), “An analysis of deforestation patterns in the extractive reserves of Acre, Amazonia from satellite imagery: A landscape ecological approach,” *International Journal of Remote Sensing*, 21, 2555–2570.

Perera, H.A., Buse, J.L. and Crow, R.T. (2007), *Forest Landscape Ecology: Transferring Knowledge to Practice*. Berlin: Springer-Verlag.

Perz, S.G., Aramburú, C., Bremner, J. (2005), “Population, land use and deforestation in the pan Amazon Basin: A comparison of Brazil, Bolivia, Colombia, Ecuador, Perú and Venezuela,” *Environment, Development and Sustainability*, 7(1), 23-49.

Perz, S.G., Brilhante, S., Brown, F., Caldas, M., Ikeda, S., Mendoza, E., Overdevest, C., Reis, V., Reyes, J. F., Rojas, D., Schmink, M., Souza, C., Walker, R., May 2008. “Road building, land use and climate change: prospects for environmental governance in the Amazon,” *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363 (1498), 1889-1895.

Perz, S.G., Caldas, M.M., Arima, E., and Walker, R.J. (2007a), “Socio-spatial process of unofficial road building in the Amazon: Socioeconomic and biophysical explanations,” *Development and Change*, 38(3), 529-551.

Perz, S.G., Overdevest, C., Arima, E.Y., Caldas, M.M., and Walker, R.J. (2007b), “Unofficial road building in the Brazillian Amazon: dilemmas and models of road governance,” *Environmental Conservation*, 34, 112-121.

Perz, S.G., Shenkin, A., Barnes, G., Cabrera, L., Carvalho, L. A., and Castillo, J. (2012), “Connectivity and resilience: A multidimensional analysis of infrastructure impacts in the southwestern Amazon,” *Social Indicators Research*, 106, 259-285.

Pfaff, A., Kerr, S., Lipper, L., Cavatassi, R., Davis, B., Hendy, J., and Sanchez-Azofeifa, G.A. (2007), “Will buying tropical forest carbon benefit the poor? Evidence from Costa Rica,” *Land Use Policy*, 24, 600–610.

Pimm, S.L. (1984), *The Balance of Nature*, Chicago: University of Chicago Press, Chicago.

Reis, S. (2008), “Analyzing land Use/Land cover changes using remote sensing and GIS in Rize, North-East Turkey,” *Sensors*, 8, 6188-6202.

Riitters, K., and Coulston, J. (2005) “Hot spots of perforated forest in the eastern united states,” *Environmental management*, 35(4), 483-492.

Riitters, K.H., Vogt, P., Soille, P., Kozak, J., and Estreguil, C. (2007), “Neutral model analysis of landscape patterns from mathematical morphology,” *Landscape Ecology*, 22, 1033-1043.

Riitters, K.H., and Wickham, J.D. (2003), “How far to the nearest road?” *Frontiers in Ecology and the Environment*, 1, 125-129.

Riitters, K.H., Wickham, J.D., O'Neill, R.V, Jones, B., and Smith, E. (2000), “Global-scale patterns of forest fragmentation,” *Conservation Ecology*, 4(2): 3.

Riitters, K.H., Wickham, J.D., O'Neill, R.V., Jones, B.K., Smith, E.R., Coulston, J.W., Wade, T.G., and Smith, J.H. (2002), “Fragmentation of continental United States forests,” *Ecosystems*, 5, 815-822.

Riitters, K.H., Wickham, J.D., and Wade, T. (2009), “An indicator of forest dynamics using a shifting landscape mosaic,” *Ecological Indicators*, 9, 107-117.

Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sorlin, S., Snyder, P. K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., and Foley, J.A. (2009), “A safe operating space for humanity,” *Nature*, 461(7263), 472–475.

Saura, S., Estreguil, C., Mouton, C. and Rodríguez-Freire, M. (2011), “Network analysis to assess landscape connectivity trends: application to European forests (1990-2000),” *Ecological Indicators*, 11, 407-416.

Scheffer, M. (2009), *Critical transitions in nature and society*. Princeton; Princeton University Press.

Scheffer, M., Bascompte, J., Brock, W.A., Brovkin, V., Carpenter, S.R., Dakos, V., Held, H., van Nes, E. H., Rietkerk, M., and Sugihara, G. (2009), “Early-warning signals for critical transitions,” *Nature*, 461(7260), 53–59.

Scheffer, M., Carpenter, C.R., Foley, J., Folke, C., and Walker, B. (2001), “Catastrophic shifts in ecosystems,” *Nature*, 413, 6856.

Scheffer, M., and Jeppesen, E. (2007), “Regime shifts in shallow lakes,” *Ecosystems*, 10(1): 1- 3.

Schmink, M., and Wood, C.H. (1992), *Contested frontiers in Amazonia*, New York: Columbia University Press.

Skole, D., and Tucker, C. (1993), "Tropical Deforestation and Habitat Fragmentation in the Amazon: Satellite Data from 1978 to 1988," *Science*, 260(5116), 1905-1910.

Soille, P. (2003), *Morphological image analysis: principles and applications*, 2nd edition. Berlin: Springer-Verlag.

Soille, P., and Vogt, P. (2009), "Morphological segmentation of binary patterns," *Pattern Recognition Letters*, 30(4), 456-459.

Southworth, J., Cumming, G.S., Marsik, M., and Binford, M. W. (2006), "Linking spatial and temporal variation at multiple scales in a heterogeneous landscape," *The Professional Geographer*, 58(4), 406-420.

Southworth, J., Marsik, M., Qiu, Y., Perz, S., Cumming, G.S., Stevens, F., Rocha, K., Duchelle, A., and Barnes, G. (2011), "Roads as drivers of change: Trajectories across the Tri-National frontier in MAP, the southwestern Amazon," *Remote Sensing*, 3, 1047-1066.

Southworth, J., and Tucker, C. (2001), "The influence of accessibility, local institutions, and socioeconomic factors on forest cover change in the mountains of western Honduras," *Mountain Research and Development*, 21(3), 276-283.

Steininger, M.K., Tucker, C.J., Townshend, J.R.G., Killeen, T.J., Desch, A., Bell, V., and Ersts, P. (2001), "Tropical deforestation in the Bolivian Amazon," *Environmental Conservation*, 28(02), 127-134.

Strunz, S. (2012), "Is conceptual vagueness an asset? Arguments from philosophy of science applied to the concept of resilience," *Ecological Economics*, 76, 112-118.

Sun J. (2013), "Exploring edge complexity in remote-sensing vegetation index imageries," Journal of Land Use Science, http://dx.doi:10.1080/1747423X.2012.756071.

Sun, J., Huang, Z., Southworth, J., and Qiu, Y. (2013), "Mapping fractality during the process of deforestation in an Amazon tri-national frontier," *Remote Sensing Letters*. http://dx.doi:10.1080/2150704X.2013.771827.

Taylor, P., Fahrig, L., Henein, K., and Merriam, G. (1993), "Connectivity is a vital element of landscape structure," *Oikos*, 68, 571-572.

Theiler, J. (1990), "Estimating fractal dimension," *Journal of the Optical society of America A*, 7, 1055–1073.

Tilman, D., and Downing, J. A. (1994), "Biodiversity and Stability in Grasslands," *Nature*, 367, 363 – 365.

Tole, L. (2006), “Measurement and management of human-induced patterns of forest fragmentation: a case study,” *Environmental Management*, 37, 788–801.

Turner, M.G. (1989), “Landscape ecology: The effect of pattern on process,” *Annual Review of Ecology and Systematics ,* 20(1), 171-197.

Turner, M.G., Gardner, R.H., and O’Neill R.V. (2001), *Landscape Ecology in the theory and practice*. Berlin: Springer-Verlag.

Urban, D., and Keitt, T. (2001), “Landscape connectivity: a graph-theoretic perspective,” *Ecology*, 82(5), 1205–1218.

van de Walle, D. (2002), “Choosing Rural Road Investments to Help Reduce Poverty,” *World Development*, *30*, 575-589.

Vogt, P., Ferrari, J.R., Lookingbill, T.R., Gardner, R.H., Riitters, K.H., and Ostapowicz, K. (2009), “Mapping functional connectivity,” *Ecological Indicators*, 9(1), 64-71.

Vogt, P., Riitters, K., Estreguil, C., Kozak, J., Wade, T., and Wickham, J. (2007a), “Mapping Spatial Patterns with Morphological Image Processing,” *Landscape Ecology*, 22(2), 171-177.

Vogt, P., Riitters, K., Iwanowski, M., Estreguil, C., Kozak, J., and Soille, P. (2007b), “Mapping landscape corridors,” *Ecological Indicators*, 7(2), 481-488.

Wade, T.G., Wickham, J.D., Zaccarelli, N., and Riitters, K.H. (2009), “A multi-scale method of mapping urban influence,” *Environmental Modelling and Software*, 24, 1252-1256.

Wadt, L.H.O., Kainer, K.A., and Gomes-Silva, D.A.P. (2005), “Population structure and nut yield of a bertholletia excelsa stand in southwestern Amazonia,” *Forest Ecology and Management*, 211, 371–384.

Walker, B. (1981), “Is succession a viable concept in African savanna ecosystems?” in *Forest succession: Concepts and application*, eds. D.C. West, H. Shugart and D.B. Botkin, New York: Springer.

Walker, B., Holling, C.S., Carpenter, S.R., and Kinzig, A. (2004), “Resilience, adaptability and transformability in social–ecological systems,” *Ecology and Society*, 9(2), 5. [online] URL: http://www.ecologyandsociety.org/vol9/iss2/art5/

Walker, B., and Salt, D. (2006), *Resilience Thinking: Sustaining Ecosystems and People in a Changing World*. Washington D.C.: Island Press.

Wear, D., Pye, J., and Riitters, K.H. (2004), “Defining conservation priorities using fragmentation forecasts,” *Ecology and Society*, 9(5): 4.

Wickham, J.D., Riitters, K.H., Wade, T.G., and Coulston, J.W. (2007), “Temporal change in forest fragmentation at multiple scales,” *Landscape Ecology*, 22: 481 - 489. http://dx.doi:10.1007/s10980-006-9054-6.

Wickham, J.D., Riitters, K.H., Wade, T.G., and Vogt, P. (2010), “A national assessment of green infrastructure and change for the conterminous United States using morphological image processing,” *Landscape and Urban Planning*, 4(3-4), 186-195.

Wiens, J.A. (2002), “Central concepts and issues of landscape ecology,” in *Applying Landscape Ecology in Biological Conservation*, eds. K.J. Gutzwiller, New York: Springer, pp. 3–21.

Wilson, J.B., and Agnew, A.D.Q. (1992), “Positive-feedback switches in plant communities,” *Advances in Ecological Research*, 33, 263-336.

Wilcox, B.A., and Murphy, D.D. (1985), “Conservation strategy: The effects of fragmentation on extinction,” *The American Naturalist*,125, 879–887.

Wu, J., and Hobbs, R. (2002), “Key issues and research priorities in landscape ecology: An idiosyncratic synthesis,” *Landscape Ecology*, 17(4), 355-365.

Wu, J., and Qi, Y. (2000), “Dealing with scale in landscape analysis: An overview Geographic,” *Information Sciences*, 6, 1-5.

Zaccarelli, N., Li, B-L., Petrosillo, I., and Zurlini, G. (2012), “Order and disorder in ecological time-series: introducing normalized spectral entropy,” *Ecological Indicators*. http://dx.doi:10.1016/j.ecolind.2011.07.008.

Zaccarelli, N., Riitters, K.H., Petrosillo, I., and Zurlini, G. (2008), “Indicating disturbance content and context for preserved areas,” *Ecological Indicators*, 8, 841-853.

Zurlini, G., Petrosillo, I., Jones, K. B., and Zaccarelli, N. (2012), “Highlighting order and disorder in social–ecological landscapes to foster adaptive capacity and sustainability,” *Landscape Ecology*. http://dx.doi: 10.1007/s10980-012-9763-y.

Zurlini, G., Riitters, K.H., Zaccarelli, N., and Petrosillo, I. (2007), “Patterns of disturbance at multiple scales in real and simulated landscapes,” *Landscape Ecology*, 22(5), 705-721.

Zurlini, G., Riitters, K.H., Zaccarelli, N., Petrosillo, I., Jones, B.K., and Rossi, L. (2006), “Disturbance patterns in a socio-ecological system at multiple scales,” *Ecological Complexity*, 3, 119–128.

Zimov, S.A., Chuprynin, V.I., Oreshko, A.P., Chapin III, F.S., Reynolds, J.F., and Chapin, M.C. (1995), “Steppe-tundra transition: An herbivore-driven biome shift at the end of the Pleistocene,” *The American Naturalist*, 146, 765-794.

BIOGRAPHICAL SKETCH

Jing Sun was born in 1982 in Tianjin, People's Republic of China and lived in Tianjin until 2001. In August 2001, he moved to Xi'an to study Geographic Information System at Northwest University, China and received B.A.Sc. in GIS in 2005.

Jing Sun entered the Master program of Human Geography at Northwest University, China in 2005. Jing worked with Xin-jun Yang and Jun Wang, two scientists who combine a strong quantitative approach with an equal devotion to focus on applied questions. Jing studied resilience theory which introduced him to a new arena and he conducted fieldwork many times in Loess Plateau region and published one of the first resilience papers in China (been cited 27 times as of April 2012). In 2008, Jing Sun completed his Master thesis on 'preliminary exploration of the resilience of social-ecological systems to drought in semi-arid areas in northwest China'. The study during this time convinced Jing that he need attend a Ph.D. program to continue resilience study.

Jing Sun entered the Ph.D. program in the Department of Geography, University of Florida to continue and expand upon his work with Jane Southworth. He received his Ph.D. from the University of Florida in the summer of 2013.