PARKS AND POLICIES: INTEGRATING GEOSPATIAL TOOLS AND MODELING TO EVALUATE CONSERVATION INTERVENTIONS IN CENTRAL INDIA

By

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To my parents

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LIST OF ABBREVIATIONS

ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
GIS	Geographic Information Science
IGBP	International Geosphere-Biosphere programme
IHDP	International Human Dimensions Programme on Global Environmental Change
NASA	National Aeronautics and Space Administration
SMD	Science Mission Directorate

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Tropical forests worldwide are undergoing rapid changes due to increasing human populations and varied land use practices. In an effort to protect these forests and their species, the number of protected areas has increased exponentially in recent decades. While the amount of money invested into conservation has been increasing along with the number of protected areas across the globe in the past few decades, few well-designed empirical studies try to show what could have happened in the absence of the conservation efforts. This study combines tools and techniques from remote sensing, GIS, and landscape ecology to evaluate conservation interventions in a tropical landscape in Central India.

The study area, namely Pench tiger reserve, Maharashtra is embedded within a matrix of other land cover types, including non-protected forests, agricultural fields, or urban areas. It is critical to monitor this landscape for changing forest dynamics as this region can host viable tiger population in presence of adequate prey-base, which in turn is directly dependent on the forest for food and cover. This study utilizes spatio-temporal forest cover transitions, and simulated landscapes assuming absence of the conservation intervention as proxies for the effectiveness of this protected area. This

study also explores the role of continuous vegetation indices, and surface temperature derived from multiple satellite images in describing and quantifying changes in forest composition and structure. Methods used in this study are robust and can be used in any ecological settings.

Findings of this study show that while national-level policies establishing protected areas play an important role in conservation, alternative approaches including favorable management policies could also be effective to extend conservation over larger areas. This study also suggests that combination of methods from multiple disciplines can be effectively used for spatio-temporal assessment of changing forest cover, structure and composition as well as empirical evaluation of conservation intervention, particularly in developing countries.

CHAPTER 1
INTRODUCTION

Outline and Scope of the Research

Earth's carbon cycle and ecosystems are being subjected to human intervention and climate changes on an unprecedented scale, both in terms of rate and extent (NASA SMD, 2007). A better understanding of the implications of these changes is required for food production, biodiversity, sustainable resource management, and the maintenance of a healthy, productive environment (Pinstrup-Anderson and Pandya-Lorch, 1998). The scientific community recognizes the importance of frequent, repeat observations from space, at both moderate and high spatial resolutions, to address the heterogeneity of living systems. In addition, research projects that integrate an understanding of the coupled human-environment system, such as the IGBP/IHDP Global Land Project, are considered an important component of global change research. This dissertation contributes to such fields of research by integrating advanced techniques from remote sensing, GIS and landscape ecology that contribute to the better understanding of landscape dynamics in human-dominated socio-ecological systems. Specifically, this dissertation documents spatial and temporal landscape patterns emerged from differential conservation strategies in a human-modified landscape through analysis of land cover conversion and modification as well as empirical evaluation of conservation interventions.

This dissertation addresses issues related to biodiversity conservation, forest management strategies, and national level policies in and around a tiger reserve in Central India, which I am using as a template for tropical ecosystems. This research is based on quantitative analysis of multi-date and multi-platform satellite images of the

study area. There are five research objectives, which along with the corresponding research questions are listed under Table 1-1.

Study Area

Pench Tiger Reserve (PTR) is situated in the Nagpur district of the northern part of the state of Maharashtra (Fig. 1-1). The forest type is southern tropical dry deciduous forest (Champion and Seth, 1968). The floral diversity is dominated by teak (*Tectona grandis*) and includes other species such as Indian laurel (*Terminalia tomentosa*), white marudah (*Terminalia arjuna*) and bamboo (*Bambusa arundinacea*). The faunal diversity includes 33 species of mammals including tiger, leopard, spotted deer and over 160 species of birds, reptiles, and fishes. Overall topography of the region is undulating, the highest peak inside the park being 583 m. Most of the annual precipitation (1050 mm) is received between June and September, with temperatures varying from 14 to 43 ˚C.

PTR, Maharashtra shares its name and the northern boundary with PTR, Madhya Pradesh (Fig. 1-1). These two reserves form together the first inter-state tiger reserve of India. However, they fall under the jurisdiction of two different states, and thus two state forest departments. PTR, Maharashtra (area: 257.2 km^2) was declared a national park by the Government of Maharashtra in 1975 and a tiger reserve by the Government of India in 1999. Over 40 villages within 10 km from the park boundary (Fig. 1-1) host significant human (mostly of *Gond* and *Korku* tribes and agriculturists by occupation) and cattle population.

The administration and management of PTR, Maharashtra comes under the state government. The forests outside the park are commercially managed by the state government (Nagpur Forest Division) and the Forest Development Corporation of

Maharashtra (FDCM) − a public sector unit. Previously the focus of the commercial management was to replace natural mixed forests with monoculture plantations of commercially important species (e.g. teak). This changed during the 1990s, with stakeholders participating in conservation efforts, such as the World Bank assisted Maharashtra Forestry Project (World Bank, 1991).

Dissertation Structure

The dissertation consists of seven chapters with five of them structured as individual publishable papers. This introductory chapter with outlines of research objectives is followed by chapter two (Paper 1) that situates this research in its wider context by synthesizing results from published quantitative or qualitative data on forest dynamics from 24 protected areas in 7 tiger ranging countries. Chapter three (Paper 2) integrates remote sensing, GIS and field observations to examine the effects of different management strategies on land cover changes in the study area. Chapter four (Paper 3) combines a cellular automata-Markov modeling approach and a counterfactual approach showing what may have happened in this particular landscape in the absence of the national-level conservation interventions. Chapter five (Paper 4) explores the utility of 2-band Enhanced Vegetation Index (EVI2) in quantifying landscape composition and configuration. Chapter six (Paper 5) compares the performance of remote sensing indices and surface temperature in differentiating forest successions. Finally, chapter seven summarizes the findings of this study and highlights the contribution of this study to land change science in general and the environmental change in central India in particular.

Table 1-1. Research objectives and research questions

Research objective	Research question	Chapter
RO1: To synthesize the current body of knowledge on forest dynamics in tiger landscapes across Asia and to situate the present study in its wider context	What is the current status of knowledge on forest dynamics trends in and around tiger reserves/wildlife sanctuaries/national parks across Asia?	Chapter two
RO2: To compare and contrast land cover changes in and outside PTR resulting from differential management strategies	Does exclusionary management of PTR result in the conservation of forest cover and/or regeneration of forest cover? Do the spatial and temporal land cover trajectories vary in areas of commercial extraction outside the park boundaries from those found within the park? Are the changes shown by land cover change trajectories also supported by the changes in NDVI values, reflecting vegetation productivity?	Chapter three
RO3: To evaluate the conservation intervention in PTR by comparing a modeled scenario with the actual landscape	What is the likely role of the conservation interventions in guiding land cover change and landscape fragmentation in and around PTR? Can spatial model be combined with counterfactual approach to empirically evaluate conservation intervention?	Chapter four
RO4: To explore the utility of satellite-derived vegetation index as an indicator of landscape heterogeneity as reflected in different land covers in Pench landscape	Can EVI2 be used as a gradient surface to represent land cover diversity in a human-modified tropical landscape? How effective are the surface metrics, in terms of describing landscape composition and configuration?	Chapter five
RO5: To assess the utility of satellite-derived thermal indices and vegetation indices in differentiating tropical forest successions	How effective are the surface temperature, emissivity, and normalized tasseled-cap indices in differentiating tropical forest successions?	Chapter six

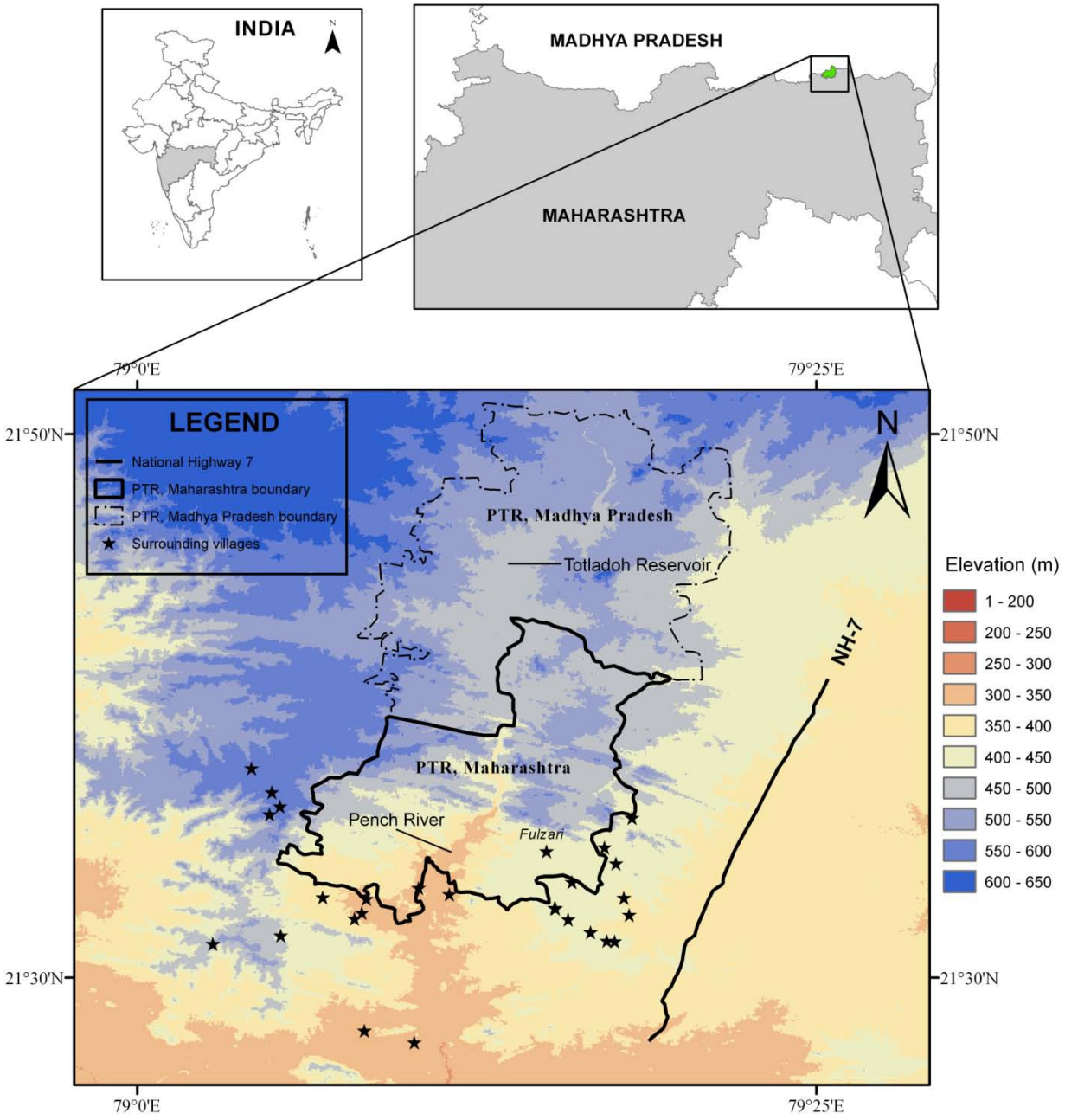


Figure 1-1. Location of the study area

CHAPTER 2
TRENDS OF FOREST DYNAMICS IN TIGER LANDSCAPES ACROSS ASIA

Introduction

The geographical range of the wild tigers has been greatly reduced over the last 50 years, with these large carnivores currently occupying a mere 7% of their historical range (Dinerstein et al., 2006). While tigers are still found in a wide variety of forest types, including the coniferous-deciduous forests of eastern Russia, the tall-grass habitats south of the Himalayas, the swamps and mangrove forests of Sundarbans, the moist/dry deciduous forests of Central Indian Highlands and the tropical rainforests of Sumatra and Malaysia, frequent reports of human-tiger conflicts are not uncommon in these heavily human-altered landscapes. One of the conservation strategies to save the wild tigers is to create tiger landscapes that are anchored on prey-rich protected areas (PA), i.e. national park/tiger reserve/wildlife sanctuary, and are linked to greater ecological systems through habitat corridors (Karanth and Stith, 1999). However, in human-dominated landscapes this strategy requires direct or indirect involvement of the local inhabitants to be effective. PAs are often surrounded by competing land uses, and have been traditional sources of livelihoods for rural people. Hence, trends in forest dynamics in a protected landscape could be used to understand the prospects for sustainable human-nature coexistence, which is a pre-requisite for long-term and effective tiger conservation.

Forest dynamics are often used to measure PA effectiveness (DeFries et al., 2005; Naughton-Treves et al., 2005; Hayes, 2006; Nagendra, 2008). Avoiding forest cover loss cannot be the ultimate test for PA effectiveness, as biodiversity can be significantly compromised by other threats, such as subsistence hunting or commercial

hunting/poaching (Redford, 1992), in an otherwise ecologically sustainable PA landscape. Nonetheless, intact or regenerated forest cover is a useful proxy for conservation success, probably because forest dynamics can be linked to many factors relevant to conservation evaluations. Forest composition, quality, and structure provide a fair representation of wild herbivore distribution as they show preferences for certain forest types (Kuijper et al., 2009; Millington et al., 2010). Additionally, forest degradation through livestock grazing suggests community dependence on the forest products, thus indicating lack of alternative means for sustainable livelihood. Intense resource competition between grazing livestock and wild herbivores might result in shrinking prey base (Madhusudan, 2004), which can have deleterious effects on tiger population (Karanth and Stith, 1999). It is also possible to measure social sustainability of a conservation effort through documenting park-people relationships, which is likely to reflect in the anthropogenic forest disturbance (e.g. destruction of forest, deliberate fire) of the region.

Forest cover, along with substantial prey occurrence and water, are the primary requirements for supporting viable tiger population and the prey base (Sunquist and Sunquist, 2002). While tigers can thrive in a wide variety of ecosystems, they have been reported to exhibit strong habitat preferences, at least for some of their activities. For example, studies have shown tiger preferences for better covered Mangrove woodlands and less disturbed grasslands in Sundarbans, Bangladesh (Khan and Chivers, 2007), less open moist-deciduous forest in Nagarahole, India (Karanth and Sunquist, 2000), less degraded or primary forest in Indonesia (Linkie et al., 2008), and mixed deciduous open forest and mixed tall grasslands in Nepal (Smith et al., 1998). Evidently, tigers

seem to prefer undisturbed, less degraded forested habitat for better cover and higher prey density. However, tigers have also shown high adaptability in selectively logged forests of Malaysia (Rayan and Mohamad, 2009) and are expected to benefit from forest type conversion in Russia (Cushman and Wallin, 2000). Nonetheless, lack of habitat corridors between isolated forest patches (or between two PAs), such as human settlements, extensive agricultural fields, and other land uses breaks the continuity of genetic exchanges between two isolated populations, eventually causing biodiversity loss (Heywood, 1995). Most importantly, forested habitats are essential in order to support prey abundance and distribution, since prey depletion is often considered the most important factor driving decline of tigers throughout their range (Karanth and Stith, 1999). One of the reasons behind the extinction of the Javan tiger is the conversion of the forested habitats of the island to teak during the 1900s, resulting in rapid declination of the prey base with eventual extinction of Javan tigers (Seidensticker, 1987). Hence, information on current status of forest cover dynamics is an important component not only in designing movement corridors and mixed-use transition zones surrounding PAs, but also to take timely decision to protect forest-dependent prey species.

Identifying trends in forest dynamics requires time series analysis of spatial patterns of forest cover. Remote sensing provides particularly useful tool in landscape-level spatial analysis by providing satellite snapshots at multiple spatial and temporal scales (DeFries et al., 2005; Nagendra, 2008). Satellite image analysis is probably the most widely used technique in global-level studies analyzing transformations in the biophysical and ecological attributes of the earth's surface (DeFries et al., 2005; Nepstad et al., 2006). However, such analyses are sometimes considered to be less

capable of capturing degradation (i.e. changes within a single land cover), which also is an important indicator of forest dynamics. Field-based ecological study can be particularly useful in such cases in identifying changes in forest structure, composition and diversity (Kumar and Shahabuddin, 2005; Yadav and Gupta, 2006; Mehta et al., 2008). Such studies are expensive and time-consuming compared to remote sensing studies; however, these methods can complement each other to provide perspectives on both forest conversion and modification.

Much attention has been given at national and international levels to establish a common platform for tiger conservation strategies in the 13 tiger ranging countries (Global Tiger Initiative, 2010). These 13 countries, namely Bangladesh, Bhutan, Cambodia, China, India, Indonesia, Laos, Malaysia, Myanmar, Nepal, Russia, Thailand, and Vietnam, represent a wide spectrum of socio-ecological factors that impact landscape dynamics. In order to propose feasible conservation strategies, the socio-ecological factors governing the spatial landscape patterns must be understood and taken into account. One way to achieve this goal is by looking at landscape dynamics and drivers, especially within and outside PAs. Here I synthesize results from published quantitative or qualitative data on forest dynamics from 24 PAs in 7 tiger ranging countries (Fig. 2-1). The trends of forest dynamics were identified as the temporal changes in forest covers/composition/density, mostly within the entire landscape in which the PA is embedded, with occasional exclusion of neighboring landscapes. Additional data from these 7 countries were used to understand the nature of the institutional regime and the proximate factors governing the landscape dynamics. This study uses both landscape-level and species-level published data on forest dynamics in

tiger landscapes to identify the dominant trend and most discussed drivers of change in these landscapes, which could further be used to locate priority landscapes for habitat restoration. I expect to see a wide spectrum of forest cover dynamics, some showing severe deforestation, with some with reforesting trend. This expectation is consistent with the wide variety in this database in terms of population density, location of the PA, and socio-economic structure of the society.

Methods

Two academic databases – Web of Science and Google Scholar – were used to locate peer-reviewed articles containing quantitative information about forest cover within and around PAs (national parks, tiger reserves or wildlife sanctuaries) with reported tiger populations in 13 tiger countries. I did not include PAs which, even though located within these countries, do not host wild tiger population. I considered the articles for analysis only if they have used remote sensing techniques or field-based ecological methods to report spatial or temporal changes in habitat quality or forest cover in general. Most of the remote sensing studies have reported forest cover change over decades, sometimes before and after protection. Some of the studies, however, only provide single-date measurements and/or maps of forest cover, and have used either satellite images or various field-based methods. These studies were included in this analysis only if they provided important information on spatial pattern of forest cover/composition/density, such as within or outside PA boundaries. Information was primarily selected from 27 journal articles from 7 countries – Bangladesh, China, India, Indonesia, Myanmar, Nepal, and Russia. While considerable amount of research has been conducted in rest of the tiger ranging countries, most of them report tiger

population density (commonly using camera-traps), and do not provide information on changes in forest cover or habitat, hence these were not included in this study.

Results

Summary Characteristics of the Case-Studies

A total of 24 case studies were found that met all the criteria for inclusion in this meta-analysis. A majority of the case studies are from India (~54%), with 3 each from China and Indonesia (~12.5%), 2 from Nepal (~8%), and 1 each from Bangladesh, Myanmar and Russia. Since socio-economic factors often vary from one country to the other, there would be some bias in the findings due to the over-representation of the Indian case studies. However, this also reflects the great number of tiger reserves in India, and emphasizes the need for such research in the tiger ranging countries that are not represented in this dataset.

The size of the study area ranges from less than 500 km^2 to over 2000 km^2 (Fig. 2-2a), with the largest covering about 7000 km^2. The institutional management regime for most of the cases is centralized, with PAs under government management, except for China and Nepal. In China, collective forests total about 60% of the total forest area, and provide legal resource extraction and management rights to the communities, in spite of being government-controlled (Miao and West, 2004). In Nepal, communities are partially involved in buffer-zone forests in some PAs (Nagendra et al., 2005). In India, some of the study areas participated in eco-development projects (Fig. 2-2b) which were expected to be participatory efforts to reduce extraction and grazing pressure from the core as well as the surrounding landscape by reducing forest dependence for fuelwood (through supply of bio-gas) and controlled grazing on marked pasture. These efforts had mixed results in meeting social or ecological goals, and are considered by

many scholars to have been a ‘failure’ to introduce a participatory management regime in inhabited landscapes (Shahabuddin, 2010).

The number of human settlements within the core of the PA varied from 0 (Bandipur, Corbett in India; Sikhote-Alin in Russia) to 27 (Sariska in India) (Fig. 2-2c). Almost all the Indian PAs host forest villages/settlements within the core. However, many Indian PAs have either relocated most of the forest villages or are in the process of relocation (e.g. Bhadra and Pench in India). The number of villages/settlements in the surrounding landscape (mostly within 10 km), however, is much higher (Fig. 2-2d). Most of the case studies have over 100 villages within 10 km from the park boundary, while only one has less than 5 villages in the vicinity of the PA (Namdapha in India).

Spectrum of Forest Dynamics: Deforestation to Reforestation

A major portion of the case studies (46%) reported forest dynamics based on multi-date remote sensing method, while 33% report forest cover status using field-based ecological methods. Only 21% of the studies employed a single-date remote sensing approach. As expected, these regions experienced very different trajectories of forest change, from degradation and deforestation to isolation of the PAs through changing land uses in the surroundings, to even reforestation in some landscapes (Fig. 2-3). A detailed description of the forest dynamics is provided in Table 2-1.

India exhibits all of the forest cover trajectories mentioned above. While degradation is prevalent in Indian PAs (Kumar and Shahabuddin, 2005; Nath et al., 2005; Karanth et al., 2006; Giriraj et al., 2008; Mehta et al., 2008), deforestation is often either dominant in the surroundings (Nagendra et al., 2006; Imam et al., 2009) or is followed by reforestation as a consequence of village relocation (Imam et al., 2009; DeFries et al., 2010) or change in degree of protection (Mondal and Southworth, 2010).

What is alarming here is that some PAs experienced deforestation within the core itself, in spite of the highly protected status (Sarma et al., 2008; Mondal and Southworth, 2010). On a positive note, PAs with maintained or regenerated forest cover are not uncommon (Singh et al., 2009; DeFries et al., 2010). However, PAs with dense to moderate forest cover in the core can be isolated if surrounded by other land uses (DeFries et al., 2010), thus resulting in restricted animal movement.

The Bangladesh case study suggests forest disturbance generated by both natural and anthropogenic factors (Giri et al., 2007). Soil erosion poses a major threat in the estuarine ecosystem of the Sundarbans, which is further aggravated by agricultural and aquacultural conversion of forested land. The study covers both the sides of Sundarbans (i.e. in India and Bangladesh) and suggests forest degradation while the net area of mangrove forests over the study period remained within the error margin. All of the Chinese and Indonesian case studies suggest severe deforestation, both within and outside the PAs (Coggins, 2000; Kinnaird et al., 2003; Linkie et al., 2004; Gaveau et al., 2007). The case study from Myanmar exhibits deforestation both within and outside the PA, with higher rates outside, followed by reforestation (Songer et al., 2009). Interestingly, the degree of protection did not appear to contribute to the forest dynamics in these case studies, as protected and non-protected forests experienced similar trajectories of forest change. On the other hand, the role that the nature of management regime plays is quite evident in the forest dynamics of Nepal. Studies have reported reforestation in previously degraded forests in Nepal, which is often linked to institutional decentralization (Nagendra et al., 2005; Panta et al., 2008). The Russian case study suggests varied degrees of disturbance in non-protected areas,

with higher disturbance in easily accessible forests (Cushman and Wallin, 2000). However, the reported change in forest type (conversion to hardwood forest) is thought to be beneficial for habitat generalist species such as Siberian tiger.

Proximate Causes of Changes in Forested Landscapes: Country-Level Issues

Given the biophysical, socioeconomic and institutional differences in the case studies, a wide range in the proximate drivers of forest change that shape these landscapes was expected. All the factors cited as probable causes behind the changes can be grouped under 7 broad categories (Fig. 2-4), consistent with other global-level studies of drivers of land use/land cover change (Lambin et al., 2001; Geist and Lambin, 2002). Table 2-2 provides a comprehensive list of proximate drivers of forest change grouped by countries.

The two most cited drivers of forest change in the Indian case studies were fuelwood/fodder extraction and grazing (Kumar and Shahabuddin, 2005; Nath et al., 2005; Karanth et al., 2006; Davidar et al., 2007; Mehta et al., 2008; Imam et al., 2009; DeFries et al., 2010). This is consistent with the Indian scenario where most of the PAs are surrounded by forest villages, and people are dependent on forest products for subsistent living and cattle-raising. Since Indian PAs are exclusionary in terms of institutional regime, forest villages are often relocated outside the PA boundary, thus exerting more pressure on the surrounding landscape (Nagendra et al., 2006). Illegal/commercial timber collection through selective logging is also not uncommon (Imam et al., 2009; Mondal and Southworth, 2010), sometimes even from the core forest. Agricultural conversion, including a shift towards tea/coffee plantations have also been cited as a common cause for clear felling (Giriraj et al., 2008). Post-independence India experienced rapid infra-structural growth, as a result of which considerable

amounts of forested regions were cleared and later submerged for hydro-electric projects (Bhat et al., 2001). Hence, controlled flooding, along with natural floods, has been mentioned as a common cause of deforestation in many Indian PAs (Nath et al., 2005; Sarma et al., 2008; Mondal and Southworth, 2010). Wildfire and road development leading to higher tourism pressure and improved accessibility to local markets are among the other causes that have been identified in the Indian case studies (Nagendra et al., 2006; DeFries et al., 2010).

Even though cited in most of the Indian case studies, fuelwood/fodder collection was not mentioned as a proximate cause of forest cover change in other countries, except Nepal (Nagendra et al., 2005). The other cause cited for Nepal was logging (Panta et al., 2008), which is also prevalent in Indonesia and Russia (Cushman and Wallin, 2000; Linkie et al., 2004). Wildfire is another common contributing factor of deforestation and/or degradation in Indonesia and Russia (Cushman and Wallin, 2000). However, the most important driving factor in Indonesia is probably agricultural conversion; as even the core forests are under direct threat of agricultural encroachments, especially along the boundaries (Kinnaird et al., 2003; Gaveau et al., 2007). Agricultural expansion and flooding were the only two causes mentioned in the case study from Myanmar (Songer et al., 2009). These two factors were also held responsible for mangrove forest degradation in Bangladesh (Giri et al., 2007), along with salinity change and shrimp farming. The three PAs from China included in this study mentioned plantation, especially bamboo forestry, as the major driver of change in forest composition within and outside the PAs (Coggins, 2000). However, it is difficult to reach any conclusion on the factors responsible for forest cover change from the very

few case studies from Bangladesh, China, Myanmar, Nepal, and Russia. The list of the factors from these countries is likely to expand with inclusion of more case studies.

Discussion

Forest Dynamics in Individual Countries: Trends, Challenges and Implications for Tiger Conservation

Bangladesh

The mangrove swamps of the Sundarbans, Bangladesh provide the last remaining refuge to the Royal Bengal Tiger. This mangrove ecosystem, shared between Bangladesh (60%) and India (40%), is the world's largest coastal wetland. Although this dynamic ecosystem frequently suffers from massive soil erosion due to its geographic location, the resulting forest loss is often offset by the accreted land and new mangrove plantations (Giri et al., 2007). In addition to the natural disturbances, mangrove forests of Bangladesh have a long history of human exploitations as well (Seidensticker et al., 1991). A vast proportion of mangrove forests was cleared and converted to agriculture during Turk and British colonization. Later, in spite of having a PA network, clear felling and timber extraction were allowed within the protected and reserved forests, leading to major mangrove forest loss in three decades since independence in 1947 (Richards, 1990). Today's densely populated Sundarbans is threatened not only by agriculture, but also rapidly growing aquaculture. Local people depend on forest products for subsistence living, and exert additional pressure in the mangrove forests by harvesting fish and shrimp/prawn larvae (Gopal and Chauhan, 2006). Besides forest degradation, human activities (such as forest clearance and conversion to other land uses) also directly influence fresh water flow and sediment accretion in this ecosystem. Such

changes in salinity can have deleterious effects on vegetation types, herbivorous species, and the predators (Hussain and Acharya, 1994).

To minimize forest loss the Government of Bangladesh adopted its second forest policy in 1994 in order to expand the PA network, to design and implement multiple usages of Sundarbans, and to involve local people through afforestation of encroached forests and other activities (GoB, 1994). However, none of the existing forest policy identified the importance of buffer zones surrounding the PAs, which are often embedded within an already degraded reserve forest (Sharma et al., 2008). While multiple commitments have been included in an amendment of the Bangladesh Wildlife (Preservation) Order to better protect/manage the forests and the species, challenges remain in terms of taking trans-boundary initiatives with India for Sundarbans management, limiting allowable resource extraction from the PAs, providing economically viable alternatives to shrimp farming, and implementing effective participatory management (Iftekhar, 2006; Chowdhury et al., 2009).

China

China is distinguished from other countries included in this study by its formalized land tenure system that started during collectivization and communization, and resulted in collective forests. The management regime of collective forests is complex and has changed frequently (Miao and West, 2004). During the post-revolution period (post-1949), China witnessed changes in the forest institutional regime, with many of its private, communal and state-owned forested land changing to elementary cooperative-managed forest in order to promote sustainable forestry (Liu, 2001). However, further amalgamation of cooperative lands into communes and associated corruption, along with the government-promoted iron and steel production program are believed to have

resulted in large scale deforestation (Liu, 2001). The post-revolution era also witnessed a rapid increase in PAs with over 2000 nationally protected nature reserves by 2004 (Xu and Melick, 2007). Park-people conflict has increased consequently, as traditional forest management practices are affected by conservation restrictions. It is often argued that the PAs will be more effective in conserving flagship species under traditional community management, rather than constantly changing, and confusing institutional regimes (Xu and Melick, 2007). However, the small size of many PAs, increasing PA isolation as a result of fragmentation, and high-level of human disturbance within the core forest are often held responsible for decline or complete absence of tiger and its prey species in South China tiger reserves (Tilson et al., 2004).

India

India, hosting one-third of the wild tiger population is a major player in the global tiger conservation scenario. With Project Tiger launched in 1973, Indian government assigned maximum protection to the tiger reserves in the country. Many national parks were promoted to the tiger reserve status. However, conservation practices in India continued to be a reflection of the colonial legacy of commercial extraction for a long time even after the independence (post-1947), resulting in alarming forest loss, especially in regions with limited protection (Bhat et al., 2001). In addition, other factors including frequent relocation of forest villages outside the PAs, national-level emphasis on agricultural expansion, and industrial activities (such as road network development, hydro-electric projects) contributed to post-independence forest degradation (Bhat et al., 2001; Nagendra et al., 2006; Mondal and Southworth, 2010). Fortunately, various remedial measures have been implemented nation-wide to restrict deforestation and degradation including complete ban on tree felling within any national park, and

implementation of plantation programs in the surrounding areas. Resource collection, grazing and hunting activities are strictly banned in the core reserves, while these activities are often restricted in the surroundings as well. The World Bank supported India Eco-development Project (1996-2004) aimed at reducing forest dependence among communities through livelihood support activities (World Bank, 1996). This ambitious project was not primarily a community-based project. However, one of the main objectives of this project was to increase opportunities for local participation in PA management. Unfortunately, it was never implemented as a participatory project and, given these flaws in implementation, many conservationists are skeptical about its impact on reducing anthropogenic pressure on the PA landscapes (Shahabuddin, 2010). As a result, many of the reserves are becoming increasingly isolated because of rapidly changing land uses in the surroundings (DeFries et al., 2010). With negligible participation from the forest-fringe communities in the park management and little benefit received by the communities, human-tiger conflict is increasingly obvious (Madhusudan, 2003; Ogra and Badola, 2008).

Indonesia

The link between Asian financial crisis and massive deforestation in biodiversity-rich and ecologically complex ecosystems of Indonesia is highly contested and is often political (Sunderlin, 1999; Robertson and van Schaik, 2001). However, this rapid deforestation is more alarming as Indonesia has an extensive system of PAs, detailed land use plans for commercial logging, and abundant donor assistance (Kinnaird et al., 2003). Multiple studies have documented legal/illegal logging, agricultural conversion by new settlers arriving through Indonesia’s transmigration program, development of oil palm and pulpwood plantations as driving factors for Indonesia’s massive deforestation

(Barr, 2001; Robertson and van Schaik, 2001; Holmes, 2002). Since tigers show a tendency to move away from forest edges to avoid disturbances, such as hunting and other human activities (Griffiths and van Schaik, 1993), only a fraction of the core forest is available to these wide-ranging species. As a result, human-tiger conflict is very high in disturbed areas where the probability of human-tiger interaction is maximum (Nyhus and Tilson, 2004). Declaration of more PAs that have suitable habitat and prey base and that can serve as corridors between isolated forested patches, such as Batang Gadis National Park (Wibisono et al., 2009), can have positive impacts on tiger conservation. However, to preserve the dwindling Sumatran tiger population, the extensive agro-forestry matrix surrounding the PAs must be wisely used (Nyhus and Tilson, 2004).

Myanmar

Myanmar's complex political history has shaped changes in protected landscapes (Aung et al., 2004). Post-independence (post-1948) Myanmar had other priorities than wildlife protection, focusing on economic and institutional changes. Post-war social unrest aggravated hunting, with no provision for wildlife habitat protection in the 1954 amendment of the Burma Wildlife Protection Act (Gutter, 2001). Following the Asian economic crisis in the mid-1980s and increasing pressure for natural resources from the neighboring countries, the severity of the threat to Myanmar's forest and wildlife became apparent, and conservation strategies began to emerge in national policies. However, tiger populations continued to decline as a result of hunting, prey depletion, habitat fragmentation, inadequate PA coverage, lack of effective management, and the flourishing trade of tiger parts in global market (Lynam et al., 2006). Myanmar's first tiger reserve, also the world's largest (Hukaung valley) was recently proposed to be

created as part of a national tiger conservation strategy (Lynam et al., 2009). While extensive surveys have confirmed tiger population in extensive and continuous forested tracks of farther north and farther south of Myanmar, it would be particularly challenging to establish a sound conservation plan, especially stopping tiger habitat loss in trans-border areas and cross-border trafficking of tiger parts (Lynam et al., 2006).

Nepal

Tigers used to range across most of lowland Nepal until settlement programs in the 1960s converted most of this region to agricultural land (Gurung, 1983). Further decline in the tiger population was prevented through several government-initiated programs, such as creation of a lowland park system, community forestry in the PA surrounding landscapes, and the Terai Arc Landscape (TAL) project (Gurung et al., 2008). These projects together contributed significantly to improve habitat quality not only within the PA but also in the neighboring landscape, to reduce anthropogenic pressure on forest resources through benefit sharing with the local communities, and to link distant lowland forests that can act as breeding habitat. As a result, the Terai arc landscape is now considered to be one of the best tiger habitats in the world (Barlow et al., 2009), which underscores the need to reestablish the dispersal corridors across this landscape (Wikramanayake et al., 2004). However, during the Maoist insurgency (2000-2007), conservation efforts were severely affected both in national parks and community-managed forests, resulting in a rapid increase in wildlife (especially rhino) poaching in and around Chitwan and Bardia national parks (Martin et al., 2008). While no study to my knowledge has identified the political instability as a direct factor affecting tiger conservation, Baral et al., (2006) have documented the adverse effects of

such political turmoil on the wildlife population, habitat, and overall conservation efforts in Nepal.

Russia

Unlike other forested areas in temperate zone, the Russian Far East (RFE) was able to maintain its primary forests until the last decades of the last century, since when political and economic changes led to urbanization and industrialization of this area (Cushman and Wallin, 2000; Kondrashov, 2004). Although wildfire shapes most of the landscape-level changes in this landscape, increasing road density is rapidly coming to constitute a severe threat to the tiger population through expanding tourism in PAs, and increasing hunting, logging in unprotected areas (Kerley et al., 2002; Kondrashov, 2004). Kerley et al., (2002) found that road access severely hinders PAs from functioning as source populations of tigers, and indicate that it would be difficult to sustain a viable tiger population with increased human access in unprotected areas. Additional pressure from local communities for forest products will only further complicate the situation. While some efforts have been made to engage indigenous communities in participatory management in PA surrounding landscapes (Shmatkov and Brigham, 2003), such efforts should focus more in areas with suitable tiger habitats.

Forest Dynamics in Tiger Landscapes: Common Needs for Tiger Conservation

PAs are established to protect biodiversity; specifically in tiger landscapes the first priority is, and should be, to provide adequate protection to this endangered charismatic species. However, irrespective of countries, conservation goals are often hindered by conflicting institutional regimes and management strategies, pressing need for alternate livelihood for forest-dependent communities, lack of infrastructure for effective PA management, and human-wildlife conflict (Table 2-2). All countries that harbor viable in-

situ tiger populations have identified restoring habitats and establishing habitat corridors as one of the priority issues for tiger conservation (Global Tiger Initiative, 2010). While maintaining the core forest is the primary objective of any conservation effort, the surrounding landscape should be given similar attention in order to maintain a greater ecosystem (DeFries et al., 2007). An intact or minimally disturbed greater ecosystem is a prerequisite to host a thriving prey base, without which it is almost impossible to support a viable tiger population (Karanth and Stith, 1999). As identified by Laurance (1999), it is not enough to conserve isolated and fragmented PAs; the quality of the peripheral forest matrix must also be protected and regenerated in order to decrease the risk of mortality for wide-ranging mammals. Unfortunately, demands for maintaining such a landscape often severely underestimates the regular needs of the local communities who live in and around tiger ranging PAs.

It has long been identified that development needs of the local communities that share the ecosystem with tiger and its prey species must be addressed to further conservation goals (MacKinnon et al., 1999). However, from the case studies discussed here, these issues are evidently not being addressed sufficiently. According to this meta-analysis, tiger reserves are often subjected to agricultural, infrastructural and commercial developments (Table 2-2), which are reflected in deforestation/degradation in the surrounding landscape, if not in the core forest (Table 2-1).

Further, almost none of the countries have a participatory management regime balancing the needs of the forest-fringe communities and the conservation goals. Decentralized management has attracted much attention over the last decades, accused by conservationists of not being “conservation-centered”, and by critics of

coercive conservation as not being sufficiently “development-oriented”. The balance between conservation and development is society specific and context specific, and a complex task that is not easily achievable, requiring sustained financial, social, institutional and political commitment at all levels from national to local. Of the countries discussed here, only Nepal has a community forest and park buffer zone management program that incorporates community participation in the forest management, albeit with negligible power to introduce/modify rules of the government authority managing PAs. While India has experimented with initiatives towards involving local communities in eco-development programs, these have been widely critiqued as being far less participatory in practice, and not having achieved their stated goals of providing local communities with a stake in conservation goals (Ostrom and Nagendra, 2006; Shahabuddin, 2010). Yet, much of the deforestation and forest fragmentation in and around tiger embedded PAs can be traced to human impact. There is a clear and imperative need to involve local forest-dependent communities as partners in conservation initiatives on the ground, not just on paper.

Conclusion

This meta-analysis synthesized published quantitative data on forest dynamics from 24 tiger-hosting protected areas from 7 countries. Findings indicate that a majority of the protected area landscapes are undergoing rapid deforestation, degradation and forest fragmentation, across all countries surveyed. While proximate causes cited as impacting forest cover change include both natural and anthropogenic factors, the drivers of change are largely human. Many of these changes can be traced to a lack of participation of the forest fringe communities in forest management and conservation, which appears critical given that most remaining tiger habitats are in areas inhabited by

poor, rural, forest-dependent communities. A critical and alarming finding of this study is the lack of information on forest dynamics in and around a majority of tiger reserves across Asia. In an era where remotely sensed information on forest cover change is easily available, and can be interpreted with reference to field data to understand changes in tiger habitat, it is disturbing to note that such information is not more widely available.

Table 2-1. Individual description of the case-studies

	Type of study	Study Period	Forest dynamics	Source§
Bangladesh				
Sundarbans National Park	Multi-temporal remote sensing study	1977-1989-2000	**Afforestation** during the first half, followed by **deforestation** in the later half – net change within margin of error. **Degradation** is common.	Giri et al., 2007
China				
3 Nature Reserves in Southeast Uplands: Meihuashan, Wuyishan, and Longxishan	Field-based ecological study	1994-1995	**Deforestation and degradation** as most of the community managed forests are being replaced by anthropogenic bamboo monoculture. Evidence of high-level human and cattle disturbance within the core of the Meihuashan reserve.	Coggins, 2000
India				
Bandipur National Park, Karnataka	Field-based ecological study	2005-2006	Differential **degradation** within the park itself. Evidence of change in species assemblages from herbaceous, large woody trees in less disturbed zones to small woody trees, more shrub species in more disturbed zones.	Mehta et al., 2008
Bhadra Wildlife Sanctuary, Karnataka	Field-based ecological study	2002	Evidence of **degradation** close to villages. Degree of disturbance depends on size of the village, or in other words human and cattle population.	Karanth et al., 2006
Chandoli National Park, Maharashtra	Single-date remote sensing study	2005	High fragmentation within the park; severe **deforestation** outside the park. Evidence of regeneration, especially in grasslands, after relocation of forest villages.	Imam et al., 2009

Table 2-1 continued

	Type of study	Study Period	Forest dynamics	Source§
Corbett Tiger Reserve, Uttarakhand	Single-date remote sensing study	2005-2006	**Dense to moderate forest cover** for a major part of the reserve.	Singh et al., 2009
Kalakad-Mundanthurai Tiger Reserve, Tamil Nadu	Multi-temporal remote sensing study	1973-1990-2004	**Degradation** in the evergreen forest; forest type is continuously being changed to the semi-evergreen forest and grassland, with most changes during pre-establishment period.	Giriraj et al., 2008
Kanha Tiger Reserve, Madhya Pradesh	Single-date remote sensing study	2007	Considerable amount of continuous surrounding forest cover, especially in higher elevation. Core forest is in better condition than buffer forest. Evidence of **reforestation** after village relocations.	Ravan et al., 2005; DeFries et al., 2010
Manas Tiger Reserve, Assam	Multi-temporal remote sensing study	1977-1998-2006	**Deforestation** within the park even after establishment; drastic reduction in alluvial grassland.	Sarma et al., 2008
Nagarhole Tiger Reserve, Karnataka	Single-date remote sensing study	2009	Good amount of **intact forest cover** within the reserve; evidence of forest corridors connecting the reserve with the surrounding. However, the reserve is embedded within a matrix of agricultural fields and tea/coffee plantation.	Mahanty, 2002; DeFries et al., 2010
Namdapha Tiger Reserve, Arunachal Pradesh	Field-based ecological study	**---**	**Degradation**, especially in surrounding areas.	Nath et al., 2005
Pench Tiger Reserve, Maharashtra	Multi-temporal remote sensing study	1977-1989-2000-2007	**Deforestation followed by reforestation**, both in the park and the surroundings.	Mondal and Southworth, 2010

Table 2-1 continued

	Type of study	Study Period	Forest dynamics	Source§
Ranthambore Tiger Reserve, Rajasthan	Single-date remote sensing study	2008	**Little forest cover outside** the park boundary; concrete wall surrounding the park and agricultural mosaic isolate the park from the surroundings.	DeFries et al., 2010
Sariska Tiger Reserve, Rajasthan	Field-based ecological study	2004	**Degradation**, mainly changes in vegetation structure and composition in disturbed areas.	Kumar and Shahabuddin, 2005; Yadav and Gupta, 2006; Shahabuddin and Kumar, 2007
Tadoba-Andhari Tiger Reserve, Maharashtra	Multi-temporal remote sensing study	1989-2001	**Reforestation/maintenance** in the reserve; primarily **deforestation/degradation** in the surroundings with reforestation at places.	Nagendra et al., 2006
Indonesia				
Bukit Barisan Selatan National Park	Multi-temporal remote sensing study	1972-1985-2002	**Deforestation** both within the park and within 10 km from the park boundary with higher rate of deforestation in the buffer, however total area of forest is higher in the park. Evidence of **reforestation** in inactive encroachments within the park.	Kinnaird et al., 2003; Gaveau et al., 2007
Gunung Raya Wildlife Sanctuary	Multi-temporal remote sensing study	1972-1985-2002	Severe **deforestation** within and outside the sanctuary with high fragmentation.	Gaveau et al., 2007
Tapan valley containing a part of Kerinci Seblat National Park	Multi-temporal remote sensing study	1985-1992-1999	**Deforestation**	Linkie et al., 2004

Table 2-1 continued

	Type of study	Study Period	Forest dynamics	Source§
Myanmar				
Chatthin Wildlife Sanctuary	Multi-temporal remote sensing study	1973-1989-1992-2001-2005	**Deforestation** both within and outside the sanctuary till 2001, with higher rates outside. Evidence of **reforestation** during 2001-2005 both within and outside the sanctuary.	Songer et al., 2009
Nepal				
Chitwan district covering Royal Chitwan National Park	Multi-temporal remote sensing study	1976-1989-2001	**Reforestation** in buffer zone forests and **deforestation** within community forests. Varying deforestation rate depending on forest types. Deteriorating forest condition outside the park.	Nagendra et al., 2004, 2005; Panta et al., 2008
Royal Bardia National Park	Field-based ecological study	2005	**Degradation** in forests closer to villages. Human disturbance impacted forest vegetation structure and diversity, specifically species richness, tree density, diversity, evenness and basal area of trees. Evidence of lack of alternative resource collection areas and livelihood need for the communities leading to more disturbances.	Thapa and Chapman 2010
Russia				
Sikhote-Alin range, covering a part of Sikhote-alinsly MAB Biosphere Reserve	Multi-temporal remote sensing study	1972-1992	**Deforestation** rate varied depending on the forest type with higher disturbance in accessible forests.	Cushman and Wallin 2000; Kerley et al., 2002

§Source includes both primary and supplementary articles used in this study

Table 2-2. Country-wise summary of proximate causes driving changes in forest structure and composition mentioned in the case-studies, and priority issues for tiger conservation

Country	Proximate causes of deforestation/degradation	Priority issues for tiger conservation§
Bangladesh	Agricultural and aquacultural conversion, flood/soil erosion, illegal/legal logging	Improving law enforcement, PA infrastructure/ management, capacity building for wildlife crime detection and habitat management, mitigating human-wildlife conflict, averting developmental threats, establishing transboundary cooperation, increasing public awareness
China	Bamboo plantations	National-level policy for wildlife protection, improving PA infrastructure and management, restricting land use changes and establishing corridor connectivity, limiting poaching, trafficking and trading tiger parts, establishing transboundary cooperation, increasing public awareness
India	Fuelwood/fodder extraction, grazing, agricultural conversion, tea/coffee plantation, illegal/legal logging, flood, fire, tourism/development	Strengthening national-level policy for wildlife protection, improving PA infrastructure and management, mitigating human-wildlife conflict, restoring habitats for tigers and prey, restricting land use changes and establishing corridor connectivity, averting developmental threats, increasing public awareness
Indonesia	Illegal/legal logging, agricultural conversion, coffee/pepper plantation, fire	Strengthening national-level policy for wildlife protection, improving wildlife crime control, improving PA infrastructure and management, mitigating human-wildlife conflict, establishing corridor connectivity, averting developmental threats, increasing public awareness
Myanmar	Agricultural conversion, plantation, flood	Improving wildlife crime control, improving PA infrastructure and management, restricting land use changes and establishing corridor connectivity, establishing transboundary cooperation, averting developmental threats, increasing public awareness
Nepal	Fuelwood/fodder extraction, illegal/legal logging	Strengthening national-level policy for wildlife protection, improving PA infrastructure and management, mitigating human-wildlife conflict, restoring habitats for tigers and prey, strengthening community-based management, establishing corridor connectivity, establishing transboundary cooperation, increasing public awareness

Table 2-2 continued

Country	Proximate causes of deforestation/degradation	Priority issues for tiger conservation§
Russia	Illegal/legal logging, fire	Establishing new PAs, eliminating gaps in institutional mandates, improving wildlife crime control, restoring prey-base, averting developmental threats, improving logging practices, encouraging community-based management, substituting forest-based livelihoods, managing forest fires

§adapted from Global Tiger Initiative website

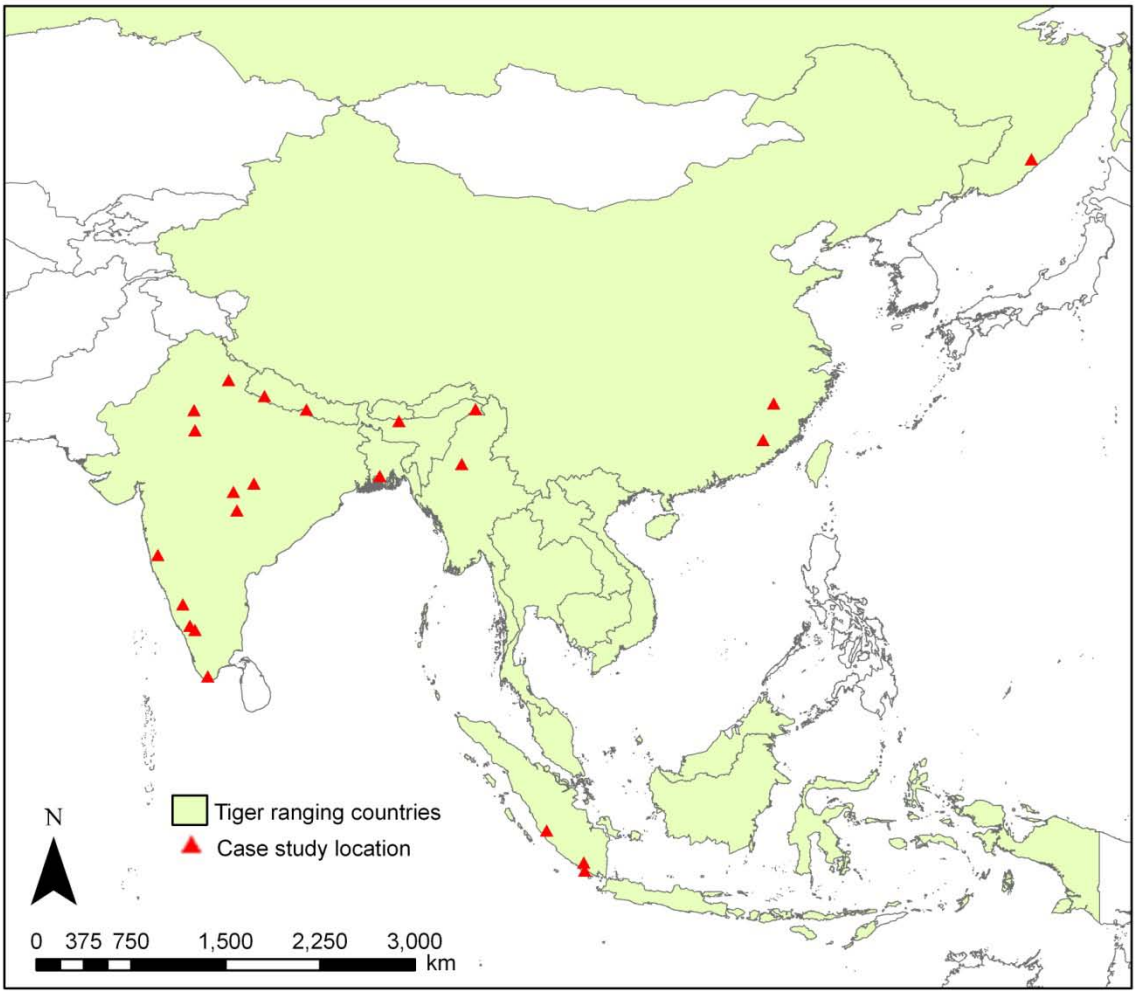


Figure 2-1. Spatial distribution of the case studies within tiger ranging countries

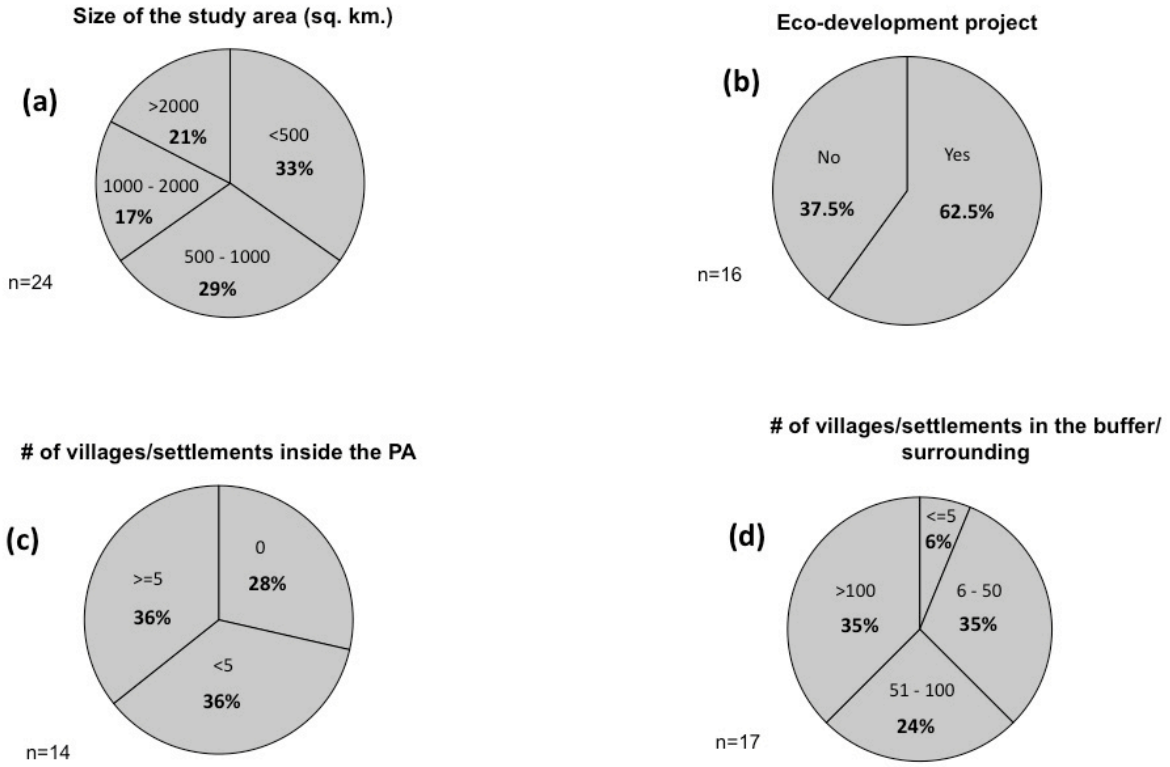


Figure 2-2. Summary characteristics of the case studies

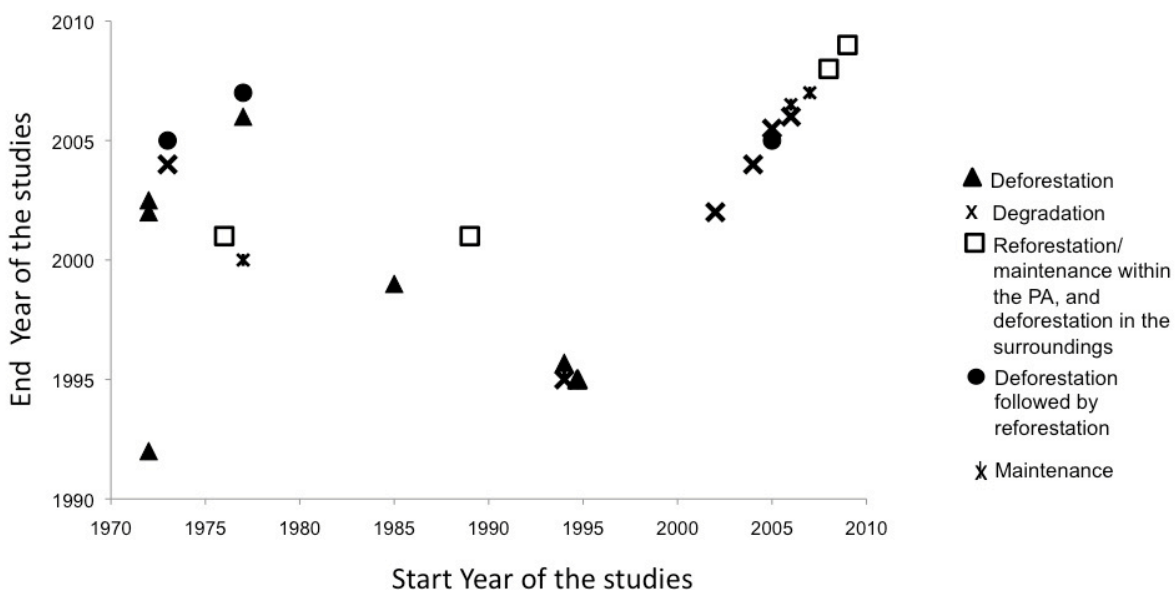


Figure 2-3. Forest dynamics and temporal span of the case studies

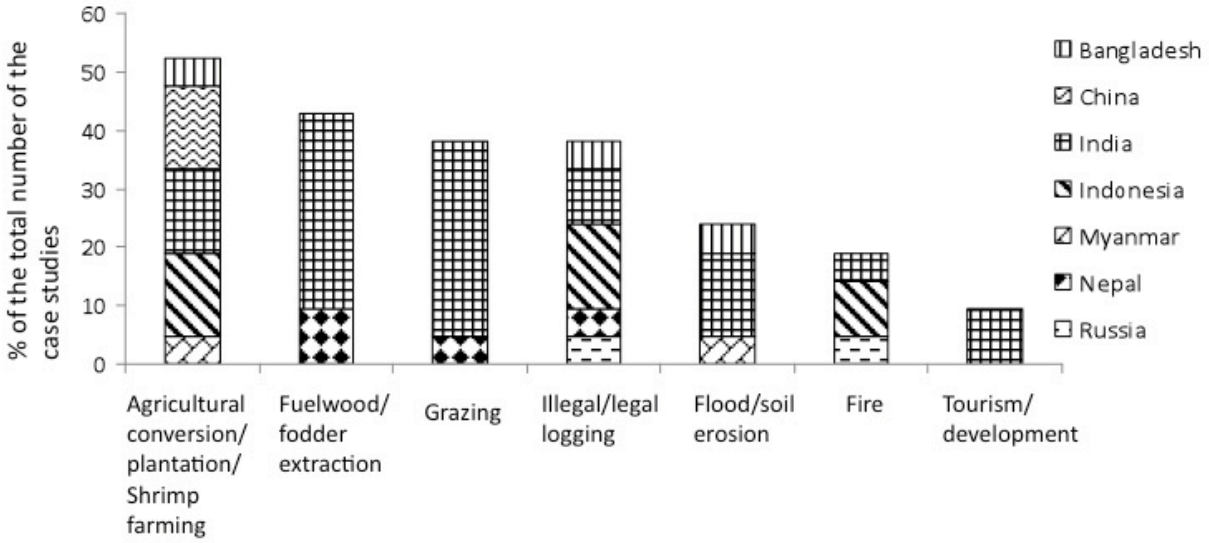


Figure 2-4. Country-wise proportion of proximate causes driving forest dynamics

CHAPTER 3
PROTECTION VS. COMMERCIAL MANAGEMENT: SPATIAL AND TEMPORAL ANALYSIS OF LAND COVER CHANGES IN THE TROPICAL FORESTS OF CENTRAL INDIA

Introduction

Human-driven land cover change is considered as the single most important variable affecting ecological systems (Vitousek, 1994) and can significantly change the amount, type and successional state of forests. These changes in forest ecosystems are considered to have global conservation importance since the effects of significant forest-cover loss or within forest modification may include massive soil erosion (Sidle et al., 2006), destabilization of watersheds (Rai and Sharma, 1998), a loss of sustainable forest uses and threats to indigenous people on a regional scale (Samal et al., 2003) and large scale release of atmospheric carbon dioxide through the burning of fossil fuels contributing more to the greenhouse effect (Houghton, 1994). Another alarming aspect of this crisis is the threat to biological diversity (Shukla et al., 1990; Laurance, 1999). Even if the target of 10-12% land protected and reserved across the globe is achieved, up to 50% of tropical species are predicted to go extinct in the next few decades (Soule and Sanjayan, 1998). While the first priority for conservation is still increasing the number of protected areas globally, it is also important to address issues related to sustainable forest management outside protected areas (Putz et al., 2001).

Parks are believed to protect forest from changes and are established to maintain carbon sinks, protect biodiversity, and to help stabilize global climate (Munroe et al., 2007). Many protected areas in tropical forests have been reported to be successful in slowing the rates of forest loss (Bruner et al., 2001). However, current controversies show skepticism in the effectiveness of parks as management regimes, which is further

complicated by the evaluation of the regional impact of parks over time (Ostrom and Nagendra, 2006). Protected areas or parks have witnessed a 500% increase in land over the past three decades (Wittemyer et al., 2008); however, most of them are under severe threat from encroachment, agriculture, ranching, urban development, illegal and legal logging, and collection of non-timber forest products (IUCN, 1999; WWF, 2004). Isolation of ecosystems due to the forest cover decline in the matrix surrounding protected areas is not uncommon in developing countries of Latin America, Asia and Africa (DeFries et al., 2005). Especially in India, where the parks frequently contain forest villages located within or outside the administrative boundary (Nagendra et al., 2006), it might be particularly challenging for the resource managers to maintain a park and its surrounding matrix as a 'whole' ecosystem.

Several studies have challenged the idea that conservation is best served by sustainable timber or forest management (Rice et al., 1997; Bowles et al., 1998) mainly based on the argument that conservation can only be served by strict protection (Bowles et al., 1998). However, with high cost for protection, pressures for profitable forest resource extraction and human population growth, sustainable forest management is often the only chance left for maintaining forest cover and biodiversity (Whitmore, 1999). Since forest management practices in many ways depend on the country's developmental stage and politically determined economic incentives (Garcia-Fernandez et al., 2008), more smaller-scale case-study type approaches are needed to understand the effect of different management strategies on forest-cover transitions.

Tropical dry forests, even though more threatened and less protected than tropical moist forests (Murphy and Lugo, 1986; Janzen, 1988), are particularly under-studied in India (Sagar and Singh, 2004). Dry forests have often been preferred by humans due to

their fertile soils (more suitable for agriculture) and more favorable climate for livestock (Murphy and Lugo, 1986). They have also been subjected to unsustainable forest resource extraction pressures (Bullock et al., 1995) to support high human population densities. Resource managers are often confronted with the daunting task of managing natural resources sustainably while meeting the needs of the human population. This can be a specific problem in developing countries (Gadgil, 1991; Rodgers, 1991), e.g. India, where approximately 50 million people depend directly on forests for their livelihoods (Hegde et al., 1996), and where substantial loss of forest area might result from continued grazing, fuelwood collection and non-timber forest product extraction (Kothari et al., 1989). More studies involving long-term monitoring of tropical dry ecosystems are thus required to develop effective forest management policies (Kumar and Shahabuddin, 2005).

This study contributes to the land change science and conservation literature by examining the effects of different management strategies on land cover changes, within and outside Pench Tiger Reserve (PTR), Maharashtra in Central India, using a host of spatial data, GIS tools, and field observations. Among different forest management practices within India (such as commercial timber extraction, participatory management, and community-based management) the present study deals with two opposing strategies − strict protection and commercial extraction. While I expect to see different forest cover change trajectories resulting from different management practices, a balance should be maintained between forest covers within and outside the park to avoid isolation. PTR, Maharashtra was chosen for this study as it is one of the reserves in the Indian subcontinent which could potentially host a large tiger population (Ranganathan et al., 2008); thus maintaining the forest cover and ecological health of

the park and its surrounding matrix can contribute to the local chapter of the global tiger conservation effort. This study uses both discrete land cover maps and continuous Normalized Difference Vegetation Index (NDVI) data sets for a spatial and temporal land cover change analysis, specifically asking the following questions:

- Does exclusionary management of PTR, Maharashtra result in the conservation of forest cover and/or regeneration of forest cover?
- Do the spatial and temporal land cover trajectories vary in areas of commercial extraction outside the park boundaries from those found within the park?
- Are the changes shown by land cover change trajectories also supported by the changes in NDVI values, reflecting vegetation productivity?

Indian Forestry—Policies and Practices

Pre- and post-independence forest management practices in India have been mostly guided by national-level policies (Bhat et al., 2001). Forest categories such as "Reserved Forest" (RF) and "Protected Forest" (PF) were first introduced during British rule (Forest Protection Act, 1927), and were retained by the Government of India after independence (post-1947). Presently, RF enjoys higher degree of protection than PF, since no hunting, and/or grazing are allowed in the former except under specific orders. Continued conservation efforts have led to pockets of RF (often upgraded to national parks and wildlife sanctuary) embedded within a PF matrix. Besides different management strategies, other factors including frequent relocation of forest villages outside protected areas, national-level emphasis on agricultural expansion, and industrial activities (such as road network development, hydro-electric projects) contributed to post-independence forest degradation (Bhat et al., 2001; Nagendra et al., 2006).

The focus of colonial-period policies was to extract commercial timber under several “working circles” (which is an area allocated to timber extraction under a “working plan” and which has a 30-year extraction cycle), which is still being practiced in non-reserve forests. Post-independence priority changes of the Indian government introduced need-based forest management where vast areas of forest were cleared and selectively cut (Pandey, 1992; Prasad, 2000) to supply timber for farm-building construction, industry, and hydro-electric projects. However, continued forest loss set off an alarm and various plantation programs were implemented (1980 onwards) to increase forested areas, although most resulted in failure (Pandey, 1992). In 1988 the forest policy was revised (MoEF, 1988) with priority given to forest and biodiversity conservation rather than financial benefits from forests and acknowledging the need of community participation in the co-management of degraded forest. As a result of this amendment, the post-1988 period witnessed a complete ban on tree felling within any national park contrary to the usual practice of controlled timber extraction through working circles within national parks. In the following two decades several other changes in government policies have continued to identify the importance of community involvement in conservation, such as the National Afforestation and Eco-development Board (NAEB) set up in 1992, participatory management programs such as “Joint Forest Management” (JFM) and “Village Eco-development” (VED), the World Bank funded India Eco-development Project (1996–2001) and the Wildlife Amendment Act (2002). While it is particularly difficult to find one ‘optimal’ management regime in the tropics (Pearce et al., 2003), comparative analyses of different regimes might be an effective tool to find regional solutions.

Methods

Satellite Imagery and GIS

A multi-temporal and multispectral dataset were used for this study (Table 3-1). While the sensors offer different spatial and spectral resolutions, such multispectral datasets are often unavoidable in studies spanning over several decades and have been successfully applied in other regions (Zoran and Anderson, 2006; Ahmed et al., 2009). The images used for this study are from the post-monsoon season (October to January) to minimize the cloud cover and phenological variations.

The park boundary along with a 10 km buffer (which is required to be treated as an eco-sensitive zone under the Environment Protection Act, 1986 and thus is required to receive strict protection) was used for this study. Multiple zones were then created starting from the park boundary for every 2 km up to the required buffer limit of 10 km, and henceforth would be mentioned as zone 1 through 5 for convenience (Fig. 3-1). I used this 2-km exclusive buffer-zone approach to understand the role of distance from the park boundary in the land cover changes. Since this study involves comparisons of land cover changes of a park and its surrounding matrix, and is not a comparison of two parks, I excluded PTR, Madhya Pradesh from the buffer zones.

Image Pre-processing

Image pre-processing was carried out in ERDAS Imagine 9.3 and ENVI 4.3 software packages. The ASTER images, being Level-1B data, were generated by applying coefficients for radiometric calibration and geometric rectification (Abrams et al., 2002). A mosaic of the two ASTER images was created to cover the study area and was then re-projected to the UTM WGS84 coordinate system (Zone 44 North) with a spatial resampling of 30 m. Geometric rectification was carried out on all the Landsat

images using the mosaicked ASTER as a basemap and nearest neighbor resampling algorithm, with root mean square (RMS) error of less than 0.5 pixels (<15 m) via image-to-image registration. Radiometric calibration and atmospheric correction were carried out to correct for sensor drift, differences due to variations in the solar angle, and atmospheric effects (Green et al., 2005).

Image Classification

A hybrid approach was used to classify the images. First, an unsupervised classification method with ISODATA clustering was used to generate preliminary classes. Then similar classes were merged based on spectral signatures and training samples collected during May‑June 2008. Finally, I defined three land cover classes – forest (both natural forests and plantations), bare soil (agricultural land and cleared forest) and water (reservoir, small ponds, and rivers) ‑ based on distinct spectral signatures and used those signatures for supervised classification with the minimum distance to means classification algorithm. Accuracy testing was performed using 143 training samples. The overall classification accuracy was over 94% with a kappa value of 0.89. Each image was classified following the same method, rather than applying signatures from one date back in time.

Change Detection

Post-classification change detection is a widely used pixel based change detection method (Jensen et al., 1995), where two (or more) classified images are compared using a change detection matrix. For this study, two-date change trajectory images (1977‑1989, 1989‑2000, and 2000‑2007) were generated for the park and for each of the buffers (Fig. 3-2). Also a four-date change trajectory image (Fig. 3-3) was generated

to capture the overall land cover changes since 1977 (1977–1989–2000–2007). In order to maintain the spatial compatibility, the classified MSS image (originally of 60 m resolution) was artificially down-scaled to 30 m before performing the change trajectory analysis.

The overall change trajectory image from four dates, with three land-cover classes for each date resulted in 81 possible change trajectories. Since interpretation of these 81 trajectories can be confusing, I further collapsed these trajectories into eight categories (Table 3-2), partly following the classification scheme suggested by Mertens and Lambin (2000).

NDVI Standard Normal Deviates and Image Differencing

NDVI was used in this study to provide estimation for vegetation productivity or biomass (Gamon et al., 1995; Sims et al., 2006). Due to the usage of a multi-sensor dataset and known correlation of precipitation and NDVI, I standardized NDVI values across dates using the following formula:

$$(NDVI_i - \mu_{NDVI})/\sigma_{NDVI}, \quad (3\text{-}1)$$

where $NDVI_i$ is the NDVI value of a particular pixel in a particular year, μ_{NDVI} and σ_{NDVI} are the mean NDVI value and standard deviation respectively for that year. These NDVI standard normal deviates record NDVI variations across an image with respect to the mean. In other words, higher NDVI standard normal deviate values show regions with higher vegetation productivity, while lower NDVI standard normal deviate values show non-forest regions.

Following the standardizations, two-date image differencing was performed to identify the changes in NDVI standard normal deviate values over time. Three ‘stable forest’ masks created from each two-date change trajectory were used to extract ‘within-

class' ecological changes in the 'stable forest' class. Fig. 3-4 shows differences between NDVI standard normal deviate values for only those areas which were forest on all dates and so in a traditional classification scheme are considered areas of 'no change'.

Results

Spatial and Temporal Land Cover Change Trajectories

The two-date change trajectories reveal a distinct spatio-temporal pattern of forest-cover changes (Fig. 3-5). Classes that converted from non-forest to forest were grouped as "reforestation" and those converted from forest to non-forest were grouped as "deforestation". As outlined in Fig. 3-5, the park suffered notable deforestation before 1989, which became negligible in later years. While the proportion of stable forest consistently increased throughout the study period, post-1989 years witnessed greater reforestation than deforestation, both within and outside the park (Fig. 3-5). A detailed description of the two-date change trajectories is provided in Table 3-3.

While the two-date change trajectories show decadal changes in forest covers, a four-date change trajectory draws an overall change scenario experienced by the study area in the last 30 years (Fig. 3-6). In addition to "stable forest", "deforestation" and "reforestation" classes, four-date change trajectory provides information on unique forest transition dynamics, namely "old forest clearing with regrowth" and "forest regrowth with new clearing" (Table 3-2), which mainly represent rotational forests showing various forest transition pathways. A detailed description of the four-date change trajectory can be found in Table 3-4.

Temporal Changes in Vegetation Productivity

Fig. 3-4 captures subtle variations in NDVI standard normal deviate values, thus in productivity or vegetation health, within so-called ‘stable’ land cover classes (forest on all dates according to traditional classification). The stable-forest NDVI standard normal deviate subtraction image for 1977‑1989 (Fig. 3-4a) suggests similar or lower vegetation productivity in the later year, leading to lower NDVI standard normal deviate values, for most parts of the study area (differences ranging from 0 to -2). During 1989‑2000, vegetation productivity seemed to increase (Fig. 3-4b), especially within the park resulting in higher NDVI standard normal deviate values in 2000 (differences ranging from 0.01 to 1). According to Fig. 3-4c, vegetation productivity in 2007 either remained same or decreased when compared to that from 2000, resulting in lower NDVI standard normal deviate values in 2007, especially within the park.

Discussion

Preceded by a global trend of tropical deforestation in the 1980s, many tropical countries witnessed a decline in deforestation rates (FAO, 1993, 2006) with a select few even exhibiting net reforestation (Lugo and Helmer, 2004; Rudel, 2005; Arroyo-Mora et al., 2005). With considerable research advances in recent years, many socioeconomic, demographic, and biotic factors have been identified promoting forest regeneration (de Jong et al., 2001; Perz and Skole, 2003). In the Indian subcontinent in particular, the expansion of forest plantations (in itself a result of earlier forest scarcity stimulating demand for forest products) have led to reforestation (Salam et al., 2000; Lamb and Gilmour, 2003). Properly planned and conducted logging is an integral part of sustainable forest management (Putz et al., 2001) and should not lead to forest-cover loss when accompanied by re-plantation or natural regeneration (Kauppi et al., 2006).

Unfortunately, commercial logging in tropical countries often represents timber ‘mining’ activity without any consideration of the renewability of natural resources (Putz et al., 2000). The present study provides a direct comparison of the effects of strict protection and commercial extraction on forest and land-cover changes and thus contributes to the forest management literature.

The study area exemplifies a typical case of contrasting management regimes. This condition enabled me to examine the role of different management approaches in preserving forest cover. While I expected to see maintained and/or regenerated forest cover within the park as a result of protection, I also expected greater forest loss over time outside the park, as suggested by many case-studies in the tropics (see DeFries et al., 2005). Part of the findings supports other studies showing the effectiveness of parks in maintaining forest cover (Bruner et al., 2001), while this study also supports the argument that commercial extraction can lead to sustainable forestry (Whitmore, 1999), at least within the time period of this study.

Indian forestry has undergone several changes from the 1980 onwards, as discussed earlier. The findings of the present study have captured the dynamics of those government policies well. While the protected status of the park became effective from 1975, it still continued to suffer notable forest loss until 1989. Especially, in the south-western boundary of the park, forested areas were cleared during 1977–1989 under “scientific management” (source: PTR management plans and personal communication with the office of the Field Director, PTR), as reflected in Fig. 3-2. Although the boundary of the park is under continued pressure from the surrounding villages (mainly grazing), this region of the park showed a forest gain in the 1989–2000

change trajectory. This pre-1989 forest loss and post-1989 forest-cover recovery within the national park could be attributed to the 1988 notification by which felling became banned in national parks, exhibiting the importance of national-level policies in favor of conservation.

The two-date change trajectory of 1977–1989 also captured the disturbances created during the Totladoh hydro-electric dam construction in the northern part of the park. A considerable amount of forested area (~20 km^2, source: PTR management plan) was submerged contributing to the forest loss during this time period. Uncontrolled forest resource extraction near the construction site also contributed to the already deteriorating forest conditions. Resettlement and relocation issues also played their part in changing forest cover and quality. At the time of notification of the park, there were two forest villages inside PTR, namely *Bodalzira* and *Fulzari*. *Bodalzira* was submerged during the construction of the Totladoh dam and the population was resettled in *Fulzari* with compensation. *Fulzari* is still located within the park (Fig. 3-1), the relocation procedure of which is in progress. The dominance of non-forest land covers in the north-western and south-eastern boundaries of the park relates to the presence of *Fulzari* and other villages in vicinity of the park which are dependent on the forest for cattle grazing and other minor forest products.

While two-date change trajectories show decadal changes in forest covers as a result of changing forest policies, a four-date change trajectory draws an overall change scenario experienced by the study area in the last 30 years. The stable forest class within the park is notably greater than that in any of the buffer zones (Fig. 3-6). With an exception in zone 2, stable forest decreases with increasing distance from the park. The stable bare soil class mainly denotes agricultural land in this area, which is much lower

within the park than the buffers (Fig. 3-6). The stable water class has negligible percentage (0.3–1.4%; refer to Table 3-3 for details), related to the temporal changes of water body locations. This might be attributed partly to the presence of the reservoir in the later images and partly to the precipitation variability resulting in more water in some images. The effect of the dam construction is also evident in the deforestation class, especially within the park (Fig. 3-3). While deforestation is considerably lower in the buffer zones, reforestation trends are quite interesting. The park gained 15.3% forest cover during the study period, which is understandable due to its protected status. However, the buffer zones which are commercially managed also show notable reforestation (20–35.2%; see supplementary material for details). It is also interesting to note that reforestation increases with increasing distance from the park boundary (Fig. 3-6).

This study supports several other studies reporting reforestation trends (Perz and Skole, 2003; Arroyo-Mora et al., 2005). However, these regenerated forests are often not ecologically equivalent to the original forest (Lugo and Helmer, 2004). The present study does not compare species level forest quality between dates; instead it uses NDVI standard normal deviate as a proxy of vegetation health and productivity. NDVI standard normal deviate image differences for the stable forest class show a decrease in vegetation productivity (Fig. 3-4) in later years in spite of an increase in forest cover suggested by discrete classification. This degradation could potentially be due to several factors, e.g. grazing, changes in tree species richness, stand density, and canopy cover as a result of plantations (Pelkey et al., 2000; Gillespie et al., 2009). While my study design does not allow me to identify the causal factors of this degradation, this

difference in forest quality is important, and will hopefully lead to more detailed ecological studies in this region.

The differences in forest coverage in and around PTR, Maharashtra most likely reflect the difference in degree of protection. Particularly, the amount of stable forest is considerably lower right outside the park (Fig. 3-5), suggesting over-exploitation and consequent degradation of the matrix. Tracts of PF outside the park are mainly managed for commercial logging and are huge sources of revenue for both the government and private sectors. These forests are also subject to severe pressure from the forest-dependent communities for cattle grazing and fuelwood collection. All of these factors might collectively contribute to the significantly lower stable forest coverage surrounding the park. However, an increasing rate of reforestation along with a decreasing rate of deforestation in the matrix is encouraging for management practices other than strict protection, particularly sustainable resource extraction. Yet, we must be cautious in giving credit solely to the commercial management practices, since a share can potentially also be attributed to, at least for the present case study, several other changes in Indian forestry as discussed below.

The India Eco-development Project (1996–2001), as mentioned earlier, aimed at involving forest-dependent communities in sustainable forest management. Among all the villages that fall within the 10 km buffer around PTR, Maharashtra only two villages, namely Sillari and Tuyapar from Maharashtra and several other villages from Madhya Pradesh were targeted under the Eco-development Project. This new participatory management regime is expected to reduce extraction and grazing pressure on the protected area as well as the surrounding buffer by reducing forest dependence for fuelwood (through supply of bio-gas) and controlled grazing on marked pasture. The

overall increase in forest cover, especially forest gain in the north-western, northeastern and south-eastern parts of the 10 km buffer, shown in the 2000–2007 change trajectory speaks for the importance of these policy changes in forest regeneration (Fig. 3-2). Yet, researchers have continually been skeptical about the degree of true participation of the communities and intention of forest officials and public sectors to engage the local people in conservation efforts (Woodman, 2004; Nayak and Berkes, 2008). Nevertheless, the concept of involving local people in conserving forest is appealing and with true implementation holds the promise of successful and potentially sustainable forest management.

Maharashtra forestry project, as mentioned earlier, was implemented with a loan from the state government and approval from the World Bank for certain models of plantation for different agro-climatic zones and species, with benefit sharing by local communities (World Bank, 1991; FDCM website). Such changes in forest management policies are well reflected in the change trajectories showing forest cover increase in the buffers over time. However, the lower percentage of stable forest right outside the park, i.e. in zone 1 (Fig. 3-5) suggests that further efforts must be made, from both government and private sectors, to engage park forest fringe communities in forest conservation without compromising their regular needs. Alternative livelihood options should be created through participatory management practices such as Eco-development Projects. A demand for such projects in the villages has also been identified (source: Management Plan of PTR, Maharashtra). It can be hoped, therefore, that with increasing awareness for conservation and reduced forest dependence through alternate livelihood options, ecological restoration of the surrounding matrix is possible.

In a landscape such as my present study area that supports scattered agriculturists, or large non-populated areas of agriculture it might not always be possible or desirable to declare a large tract of forest as a protected area. Forest resource management in such a landscape involves complex dynamics of resource economy, rural livelihoods, and conservation initiatives. While it is challenging to find a balance, stakeholders managing the surrounding matrix play a major role in limiting a park's isolation and facilitating connectivity through effective forest management.

Conclusion

This study investigates the patterns and processes of land- and forest-cover change trajectories in and around PTR, Maharashtra. It targets a landscape with different management regimes, which is representative of numerous similar scenarios in developing countries, as well as elsewhere in India. Remote sensing and GIS techniques were used to map and quantify land cover conversions. Standardized NDVI was also used to detect temporal changes in vegetation productivity.

India, as with many other countries in Asia, has witnessed forest regrowth patterns in last two decades. In the presence of many management regimes it might be particularly difficult to identify a single 'optimal' strategy. While many argue for strict protection to conserve biodiversity, proponents for sustainable commercial extraction are no less vocal. This study identifies the importance of protection as a measure of forest conservation, and also acknowledges the capability of achieving conservation goals through other participatory management practices. In the face of increasing human population growth and consequent need for resource extraction (both for personal and commercial profit) in developing countries, this research leads to a recommendation to maintain a balance between both strict protection and sustainable

extraction. That this study hopefully will provide insight to local forest resource managers interested in developing ecologically sustainable forest management strategies that can be applied to other tropical settings as well.

Table 3-1. Description of satellite imagery used in the study

Sensor Type	Acquisition date	Path/Row	Data source
Landsat MSS	January 1977	155/45	Global Land Cover Facility (GLCF) website hosted by University of Maryland
Landsat TM	November 1989	144/45	Global Land Cover Facility (GLCF) website hosted by University of Maryland
Landsat ETM+	November 2000	144/45	Global Land Cover Facility (GLCF) website hosted by University of Maryland
ASTER	October 2007	144/45	Land Processes Distributed Active Archive Center (LP DAAC) located at USGS Earth Resources Observation and Science (EROS) Center

Table 3-2. Change trajectory classes created for the four-date change analysis

Land cover categories	Description
Stable forest	Forest on all four dates
Stable bare soil	Bare soil on all four dates
Stable water	Water on all four dates
Reforestation	Non--forest[§] in 1977 and forest in 2007 irrespective of 1989 and 2000
Deforestation	Forest in 1977 and non-forest in 2007 irrespective of 1989 and 2000
Old forest clearing with regrowth	Forest in 1977 and 2007, but non-forest in 1989 or 2000 or both
Forest regrowth with new clearing	Non-forest in 1977 and 2007, but forest in 1989 or 2000 or both
Stable non-forest	Non-forest on all four dates, but changed from bare soil to water or vice versa

[§] Non-forest = bare soil or water

Table 3-3. Two-date land cover change trajectories for park and buffers[§] with area in km^2 and proportion of total area affected by each trajectory (%)

Land cover categories	Park	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5
(a) 1977-1989						
Stable forest	149.7 (60.0%)	51.4 (37.8%)	62.9 (43.4%)	63.1 (37.8%)	64.9 (34.8%)	72.1 (32.9%)
Stable bare soil	19.5 (7.8%)	45.7 (33.5%)	44.5 (30.7%)	56.6 (33.9%)	74.2 (39.8%)	94.5 (43.2%)
Stable water	1.9 (<1.0%)	1.6 (1.2%)	1.0 (<1.0%)	1.2 (<1.0%)	2.7 (1.5%)	0.7 (<1.0%)
Forest to bare soil	41.1 (16.5%)	21.4 (15.7%)	14.9 (10.3%)	21.2 (12.7%)	16.6 (8.9%)	19.4 (8.9%)
Forest to water	5.6 (2.2%)	0.1 (<1.0%)	--	0.1 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)
Bare soil to forest	9.9 (4.0%)	9.3 (6.8%)	13.9 (9.6%)	18.8 (11.3%)	21.4 (11.4%)	25.9 (11.8%)
Bare soil to water	3.2 (1.3%)	0.1 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)	0.2 (<1.0%)	0.2 (<1.0%)
Water to forest	16.4 (6.6%)	4.1 (3.0%)	4.8 (3.3%)	4.1 (2.4%)	4.5 (2.4%)	4.6 (2.1%)
Water to bare soil	2.3 (<1.0%)	2.5 (1.9%)	2.9 (2.0%)	1.9 (1.2%)	2.0 (1.1%)	1.5 (<1.0%)
(b) 1989-2000						
Stable forest	166.4 (66.7%)	56.6 (41.5%)	72.1 (49.8%)	72.4 (43.3%)	76.1 (40.8%)	84.4 (38.7%)
Stable bare soil	22.6 (9.1%)	46.7 (34.3%)	41.0 (28.3%)	55.2 (33.0%)	66.7 (35.7%)	89.0 (40.8%)
Stable water	8.4 (3.4%)	1.7 (1.2%)	1.0 (<1.0%)	1.3 (<1.0%)	2.9 (1.6%)	0.9 (<1.0%)
Forest to bare soil	8.3 (3.3%)	7.7 (5.7%)	9.1 (6.3%)	13.1 (7.9%)	14.0 (7.5%)	17.5 (8.0%)
Forest to water	0.9 (<1.0%)	0.7 (<1.0%)	0.5 (<1.0%)	0.5 (<1.0%)	0.6 (<1.0%)	0.4 (<1.0%)
Bare soil to forest	39.6 (15.9%)	21.6 (15.9%)	20.1 (13.9%)	23.6 (14.1%)	23.6 (12.6%)	25.5 (11.7%)
Bare soil to water	0.7 (<1.0%)	1.3 (<1.0%)	0.9 (<1.0%)	0.9 (<1.0%)	2.7 (1.4%)	0.5 (<1.0%)
Water to forest	0.3 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)
Water to bare soil	2.2 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)	0.1 (<1.0%)
(c) 2000-2007						
Stable forest	203.0 (81.4%)	70.5 (51.8%)	81.4 (56.2%)	87.7 (52.5%)	90.9 (48.7%)	101.1 (46.2%)
Stable bare soil	7.1 (2.9%)	29.8 (21.8%)	26.3 (18.2%)	27.7 (16.6%)	32.9 (17.6%)	42.9 (19.6%)
Stable water	9.1 (3.7%)	3.3 (2.4%)	1.7 (1.2%)	2.2 (1.3%)	5.6 (3.0%)	1.3 (<1.0%)
Forest to bare soil	2.2 (<1.0%)	7.1 (5.2%)	10.4 (7.2%)	7.8 (4.7%)	8.0 (4.3%)	8.5 (3.9%)

Table 3-3.continued

Land cover categories	Park	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5
Forest to water	1.2 (<1.0%)	0.6 (<1.0%)	0.4 (<1.0%)	0.6 (<1.0%)	0.9 (<1.0%)	0.6 (<1.0%)
Bare soil to forest	14.8 (5.9%)	23.9 (17.6%)	23.2 (16.0%)	39.3 (23.5%)	46.1 (24.7%)	62.7 (28.6%)
Bare soil to water	11.2 (4.5%)	0.8 (<1.0%)	0.7 (<1.0%)	1.4 (<1.0%)	1.8 (<1.0%)	1.2 (<1.0%)
Water to forest	0.7 (<1.0%)	0.1 (<1.0%)	0.2 (<1.0%)	0.2 (<1.0%)	0.3 (<1.0%)	0.2 (<1.0%)
Water to bare soil	0.1 (<1.0%)	0.2 (<1.0%)	0.5 (<1.0%)	0.3 (<1.0%)	0.3 (<1.0%)	0.3 (<1.0%)

[§] Zone 1 = 0-2 km, total area = 136.2 km2; Zone 2 = 2-4 km, total area = 144.8 km2; Zone 3 = 4-6 km, total area = 167.1 km2; Zone 4 = 6-8 km, total area = 186.7 km2; Zone 5 = 8-10 km; total area = 218.9 km2. Distances of the buffers are calculated from the park boundary.

Table 3-4. Four-date land cover change trajectories for park and surrounding buffers with area in km^2 and proportion of total area affected by each trajectory (%)

Land cover categories	Park	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5
Stable forest	141.8 (56.9%)	46.7 (34.3%)	57.1 (39.5%)	55.5 (33.2%)	57.2 (30.6%)	63.0 (28.9%)
Stable bare soil	4.8 (1.9%)	24.0 (17.6%)	21.3 (14.7%)	22.7 (13.6%)	28.1 (15.1%)	37.9 (17.4%)
Stable water	1.4 (0.6%)	1.5 (1.1%)	1.0 (0.7%)	1.1 (0.7%)	2.7 (1.4%)	0.6 (0.3%)
Reforestation	38.1 (15.3%)	27.2 (20%)	30.7 (21.2%)	47.3 (28.3%)	60.7 (32.5%)	76.8 (35.2%)
Deforestation	15.5 (6.2%)	5.6 (4.1%)	3.6 (2.5%)	4.6 (2.7%)	5.1 (2.8%)	4.8 (2.2%)
Old forest clearing with regrowth	38.5 (15.5%)	20.6 (15.1%)	16.9 (11.7%)	24.3 (14.6%)	19.3 (10.4%)	23.6 (10.8%)
Forest regrowth with new clearing	3.1 (1.2%)	8.4 (6.1%)	12.1 (8.4%)	9.4 (5.6%)	9.3 (5%)	9.7 (4.4%)
Stable non-forest	6.1 (2.4%)	2.2 (1.6%)	2.1 (1.5%)	2.1 (1.3%)	4.3 (2.3%)	1.8 (0.8%)

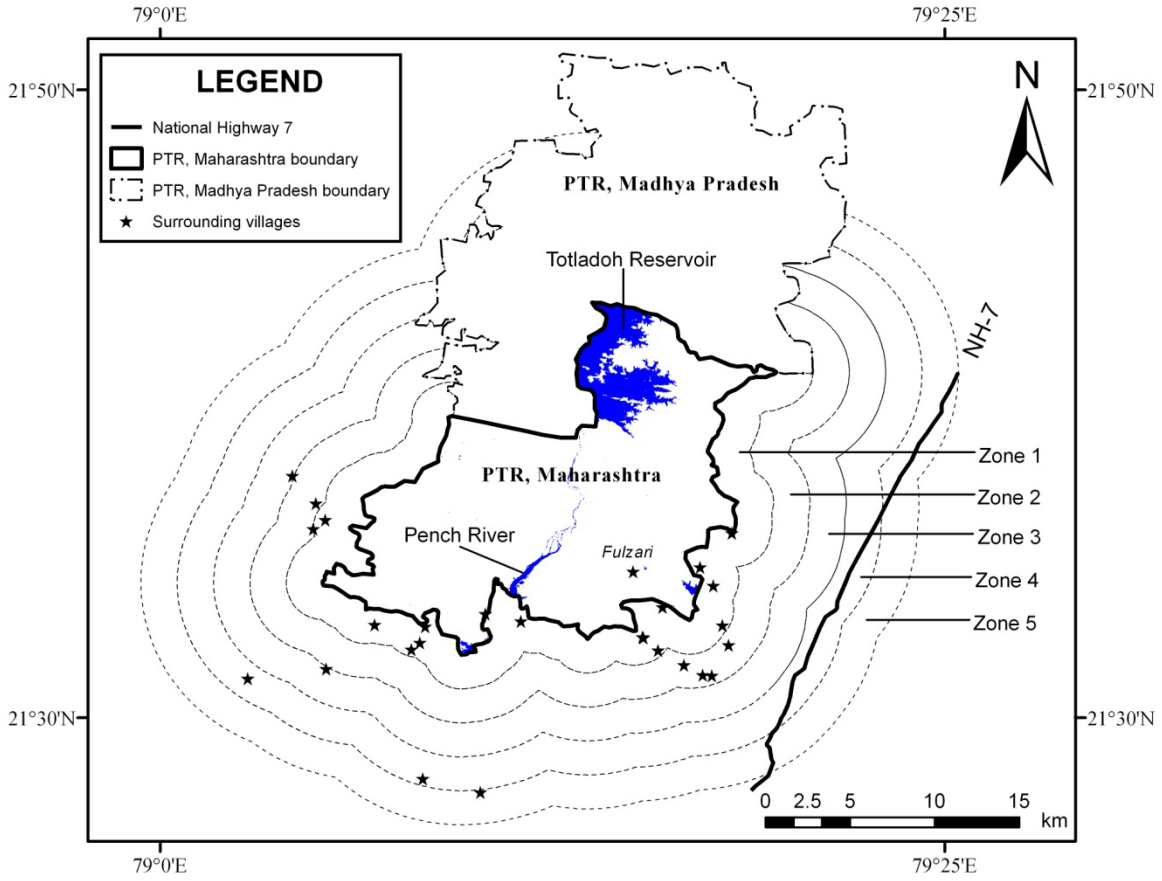


Figure 3-1. Study area showing the boundary of the Pench Tiger Reserve, Maharashtra and the buffer zones created for this study (from the park boundary up to 10 km), and locations of the visited villages.

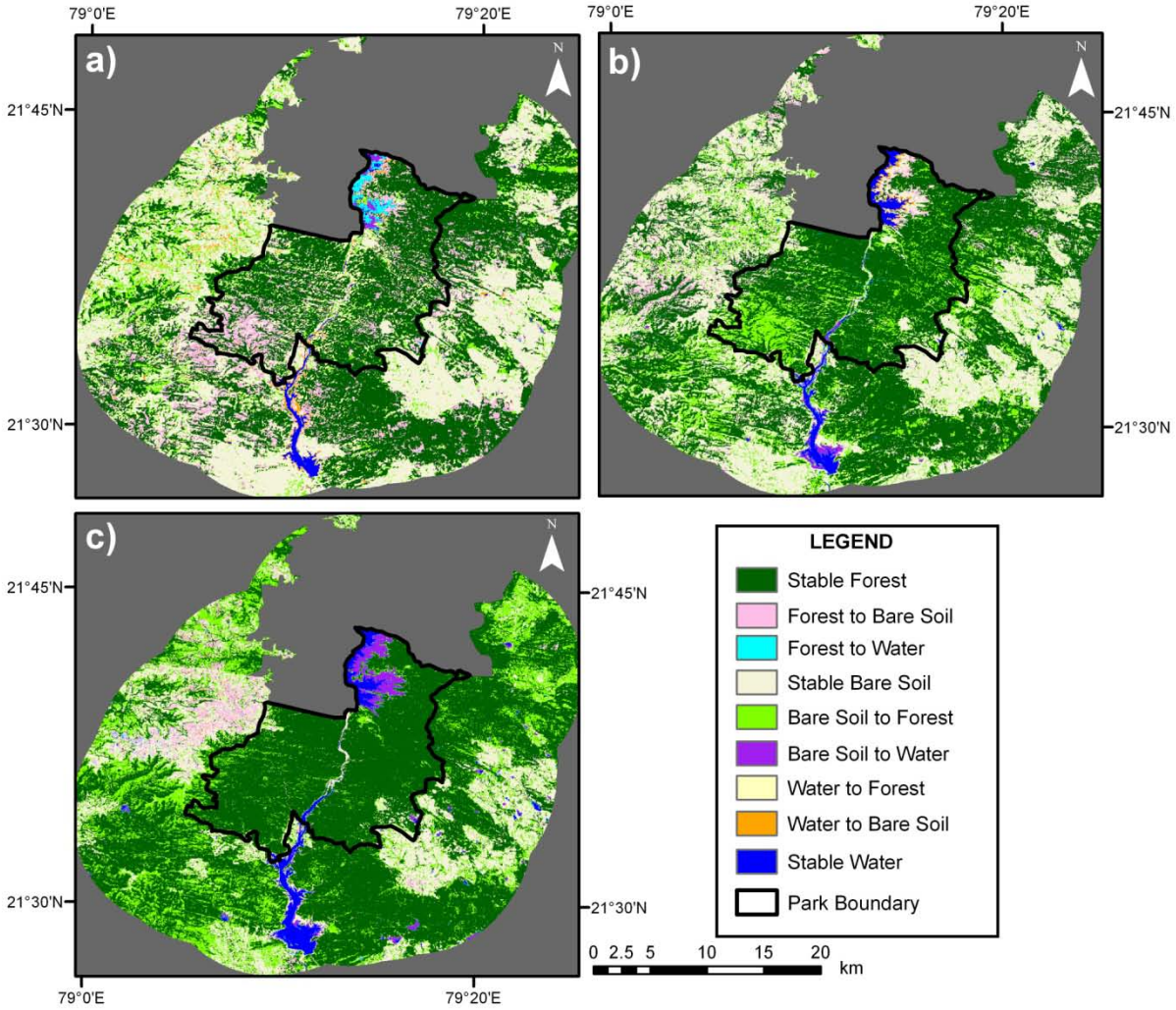


Figure 3-2. Series of two-date change trajectories of the park and 10 km buffer (a) 1977–1989, (b) 1989–2000, and (c) 2000–2007.

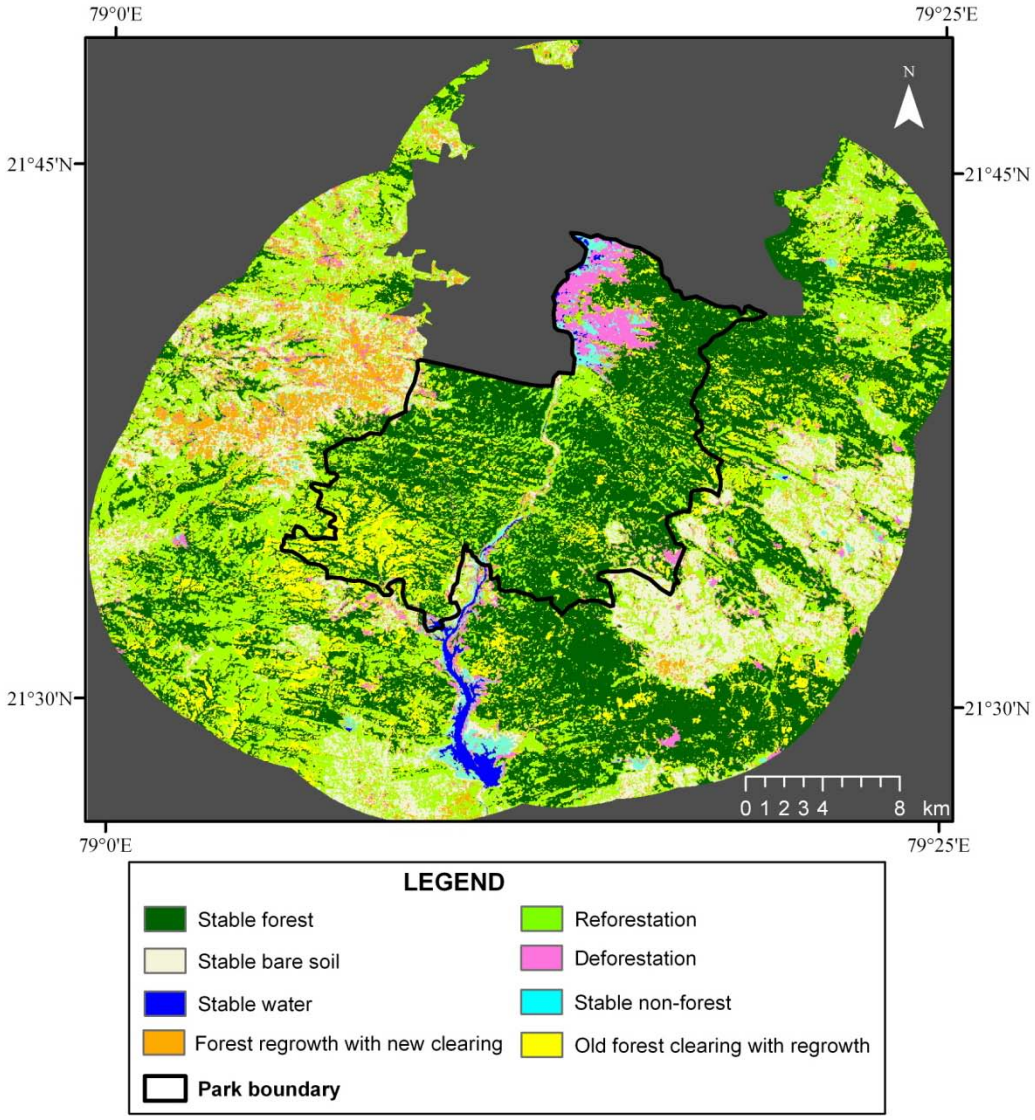


Figure 3-3. Four-date change trajectory (1977-1989-2000-2007) of the park and 10 km buffer.

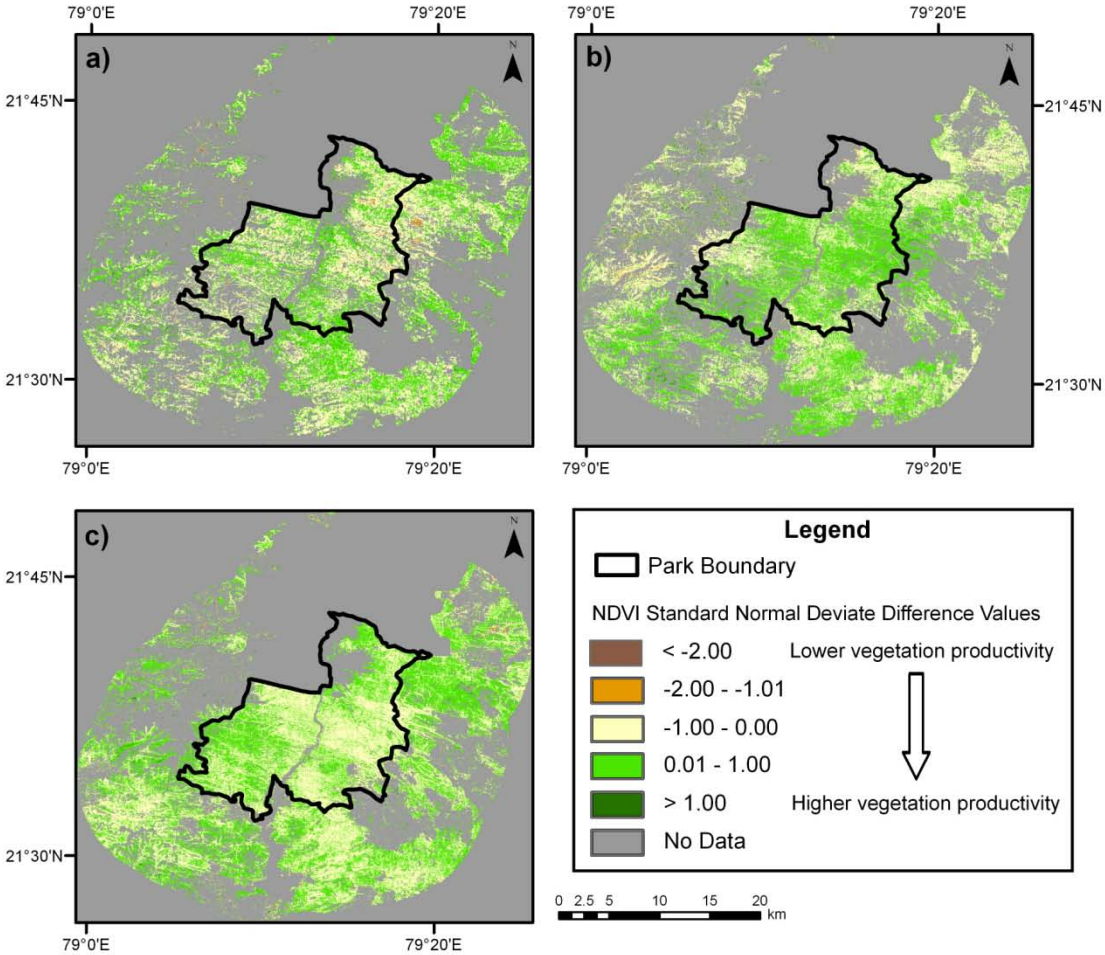


Figure 3-4. Series of two-date NDVI standard normal deviate image subtractions (color-coded) showing 'within-class' changes in NDVI standard normal deviate values for "stable forest" class (a) 1977-1989, (b) 1989-2000, and (c) 2000-2007.

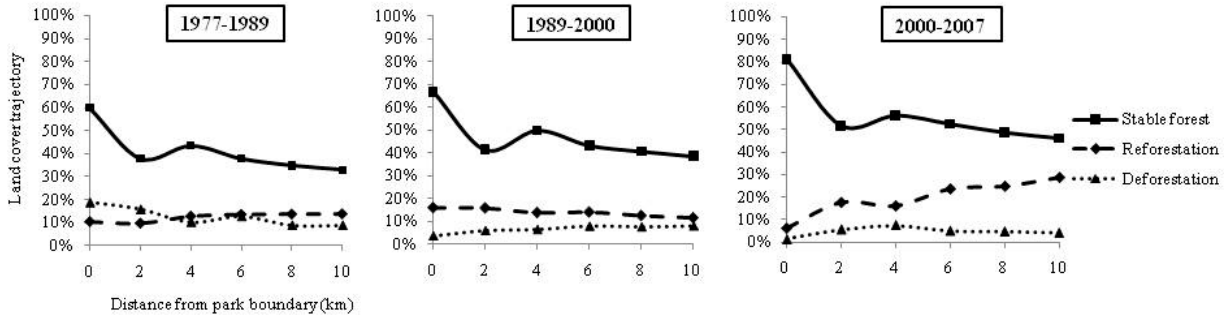


Figure 3-5. Two-date change trajectories of forest cover within the park and 10 km buffer showing changes in forest-cover transitions related to distance from the park boundary.

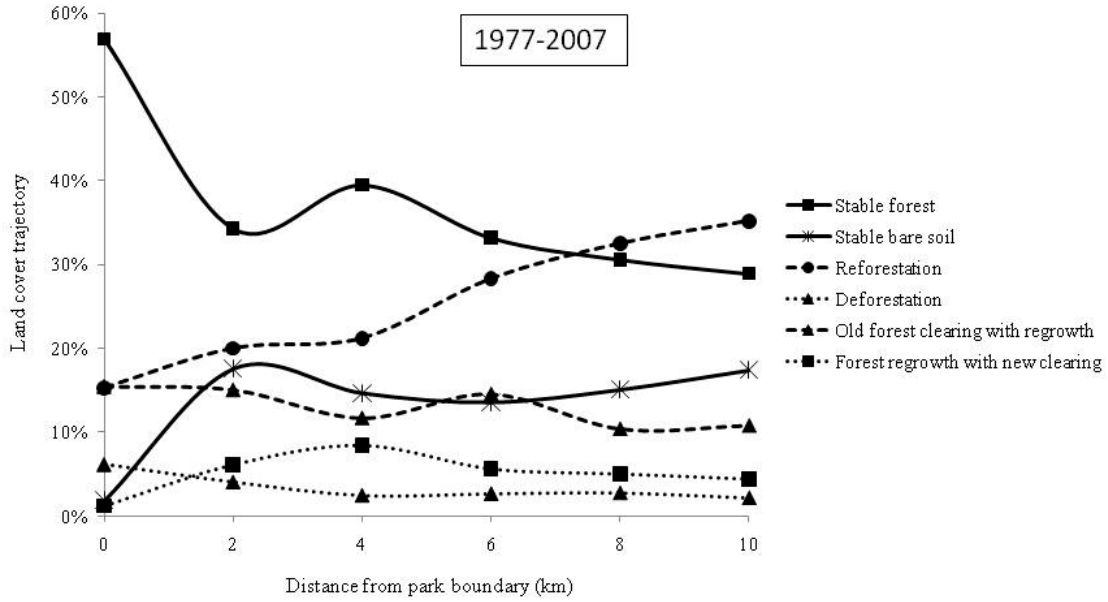


Figure 3-6. Spatial and temporal changes in land cover dynamics within the park and 10 km buffer in last three decades (1977–2007). Only classes showing notable changes (>3%) are included.

CHAPTER 4
EVALUATION OF CONSERVATION INTERVENTIONS USING A CELLULAR AUTOMATA-MARKOV MODEL

Introduction

Global environmental change has many interacting components, land use/land cover change probably representing the single most important factor affecting ecological systems (Vitousek, 1994). Changes in forest ecosystems particularly, are of global conservation importance since deforestation or degradation can have several ecological impacts, including massive soil erosion (Sidle et al., 2006), destabilization of watersheds (Rai and Sharma, 1998), large scale release of atmospheric carbon dioxide through the burning of fossil fuels contributing to the greenhouse effect (Houghton, 1994), and decline of biodiversity (Shukla et al., 1990; Laurance, 1999). Protected areas or parks are established globally to conserve ecosystems, maintain carbon sinks, protect biodiversity, and to help stabilize global climate (Munroe et al., 2007). However, current controversies show skepticism in the effectiveness of parks as management regimes, which is further complicated by the evaluation of the regional impact of parks over time (Ostrom and Nagendra, 2006) and a lack of well-designed empirical studies to estimate the effectiveness of conservation intervention (Ferraro and Pattanayak, 2006).

Positive conclusions on protected area effectiveness have been derived based on improved ecosystem conditions inside the protected areas compared to outside (Sanchez-Azofeifa et al., 2003) or perceptions of success by protected area managers (Bruner et al., 2001). However protected areas are often established in regions with minimum ecological perturbations, and hence would most likely experience little degradation over time even without the protection (Green and Sussman, 1990). Proper evaluation of conservation initiatives, therefore, should estimate what protected areas

would look like without the policy/law enforcement (Ferraro and Pattanayak, 2006). Conservation practitioners and policymakers increasingly recognize the importance of empirical evaluations of their interventions (Balmford et al., 2005). As suggested by Ferraro and Pattanayak (2006), a counterfactual approach asking a fundamental question “what would have happened had there been no intervention?” could be effective in this matter. A “counterfactual scenario” is little different from a “predicted scenario”, since for the latter it is possible to observe the scenario at the predicted time step and validate the model, but for the former the scenario is non-existing and thus it is not possible to directly compare the modeled scenario with the actual one. A conventional counterfactual approach generally compares “treatment” and “control” units which are affected and unaffected by a certain policy intervention respectively, since it is not possible in reality to know what would have happened without a certain policy. However in the real world such “units” are often unavailable because of other confounding factors simultaneously affecting one or the other unit.

In this paper, I proposed a cellular automata-Markov Chain model (CA-Markov) as an alternative to simulate a counterfactual scenario. A Markov chain model is commonly used to quantify transition probabilities of multiple land cover categories from discrete time steps. These probabilities are then used with a CA model to predict spatially explicit changes over a certain period of time. The footstone of a CA-Markov model is an initial distribution and a transition matrix, which assumes that the drivers that produce the detectable patterns of land cover categories will continue to act in the future as they had been in the past (Briassoulis, 2000). This very assumption makes a CA-Markov model suitable for a counterfactual approach, since I was interested to

extrapolate the pre-intervention landscape into the future assuming no change in the form of intervention.

A combination of remote sensing and GIS has been utilized effectively both for the CA-Markov modeling approach (Peterson et al., 2009) and the counterfactual approach (Gaveau et al., 2009). The present study integrates these two approaches using remote sensing and GIS techniques. The availability of pre- and post-intervention satellite images makes this tool particularly useful for the present study. Integrating these approaches would be ideal for study areas from developing countries, such as India, where land use land cover change around protected areas often reflect a complex interaction between law enforcement, local livelihood and different forest management activities, thus making it difficult to adapt a conventional counterfactual approach.

India provides an ideal example of a country under increasing demographic pressure and economic growth. It is also a country with mixed conservation legacy, resulting from almost 200 years of British rule. The colonial legacy of centralized forest management is clearly demonstrated by the major post-independence national-level acts/policies (Table 4-1) which have been identified as playing a pivotal role in Indian forestry and conservation (Kulkarni, 1987; Ghate, 1992; Bhat et al., 2001). However, there have been several attempts made by the Indian Government to protect forests for wildlife and local communities, rather than using them just for revenue generation, as evidenced by the forest acts/policies from the 1970s and 1980s (Table 4-1). Indian forestry, as a result, is often guided by complex interactions between national-level bureaucracies, state-level interpretations of national-level policies, local livelihoods, conservation efforts of non-governmental organizations, and commercial interests of

private sector units and state governments (Woodman, 2004; Barve et al., 2005; Karanth et al., 2006; Robbins et al., 2006; Robbins et al., 2007; Mondal and Southworth, 2010). While the number of protected areas is growing fast, it covers less than 5% of the total land area in India (National Wildlife Database, 2009). Beyond the protected area boundaries, the landscapes are typically co-managed by several institutions and are often subjected to park-focused integrated conservation and rural development efforts (Woodman, 2004). State forestry projects and eco-development projects, funded by both national and international agencies, are among such efforts that promote forest regeneration, ecological restoration and eco-development activities, especially in degraded forests surrounding protected areas. It is therefore difficult to choose “control” and “treatment” units, since “control” units, even though apparently unaffected by the protected area establishment, are subjected to other, simultaneous, conservation efforts.

Like most of the protected areas in India, PTR was subjected to various national-level policies/acts throughout the study period (1977- 2007). PTR was declared a national park by the state government in 1975 and later a tiger reserve by the central government in 1999. While many conservation efforts (both national and international) govern the landscape dynamics in and around PTR (Mondal and Southworth, 2010), the Revised Forest Policy in 1988 (an amendment to the National Forest Policy, 1980) has been identified by the academics (Ghate, 1992) and park managers (the Office of the Field Director, personal communication; PTR management plans) as probably the most important one for the study area. This policy was notably different from its predecessors as it banned tree felling in any national park in India and also acknowledged community

involvement in protection and regeneration of forests surrounding protected areas. As suggested by the park managers (the Office of the Field Director, personal communication) this policy increased the degree of protection within PTR, resulting in increased forest coverage (Mondal and Southworth, 2010). I chose the 1988 Forest Policy as the "conservation intervention" in my study as I was interested to see the changing patterns of forest transitions within the park in the absence of this intervention. Adapting the conventional "control- treatment" approach is not feasible for this study, since the implementation times of the Revised Forest Policy (1988) and afforestation and plantation strategies in the surroundings (early 1990s) overlap. While this overlapping makes it difficult to choose ecologically similar areas from the surroundings as "control" units for a counterfactual approach, it also provides a unique opportunity to generate a counterfactual scenario for the buffer as well in absence of the afforestation and plantation strategies. Here I adapt a CA-Markov modeling approach as an alternative to the conventional counterfactual approach to estimate the landscape level dynamics in absence of the conservation initiatives both for the park and the surrounding landscape.

The research questions addressed in this paper are:

- Can CA-Markov model be combined with counterfactual approach to empirically evaluate conservation intervention?
- What is the likely role of the conservation interventions in guiding (a) land cover change and (b) landscape fragmentation in and around protected areas?

Specifically I used classified satellite images prior to the 1988 Forest Policy to predict the probability of changing forest covers and fragmentation patterns assuming no intervention afterwards and then compared the predicted results to actual scenarios as an effect of the intervention. I concluded with discussing the importance of national-

level policies in forest conservation in the landscape under study and the ability of the proposed method to evaluate conservation intervention in any other socio-ecological setting.

Methods

Land Cover Maps

I used Landsat images from 1977, 1989 and 2000 and a mosaic of ASTER images from 2007 to generate land cover maps of the park and surrounding buffer areas up to 10 km (Fig. 4-1). The 1989 image was considered to represent pre-1988 policy change scenario because of the lag effect of policy creation and implementations to its effects being “seen” on the ground. PTR, Madhya Pradesh (which is a part of the 10 km buffer) was excluded from the study, since it has a different degree of protection than the rest of the buffer. Classification accuracy of the most recent land cover map is approximately 94% and was determined using training samples collected during summer, 2008. Three land cover classes were defined in the final maps – forest (both natural forests and plantations), bare soil (bare agricultural land and cleared forest) and water (reservoir, small ponds, and rivers). For this study, two-date change trajectory images for 1989-2000 and 1989- 2007 were generated for the park and the buffer in order to compare them with the modeled predicted changes in the absence of the intervention. These land cover change trajectories show areas of stable forest, stable non-forest, deforestation and reforestation, across the time periods studied.

CA-Markov Model

A Markov chain is a stochastic process that consists of a finite number of states of a system in discrete time steps and some known transition probabilities p_{ij}, where p_{ij} is the probability of that particular system moving from time step i to time step j. For

example, for a system composed of multiple land covers, the state of a particular cell at time step i denotes the type of land cover of that particular cell at time step i, which might change (or remain the same) in the next time step j. The transition probability denotes the probability of each class changing to every other class (or remaining the same) from time step i to time step j. With Markov chain analysis, future land cover can be modeled on the basis of the preceding state; that is, a matrix of actual transition probabilities between states can be used to predict future changes in the landscape from current patterns (Brown et al., 2000).

Cellular automata is a dynamic and spatially explicit modeling approach that encompasses five components – (a) a space composed of discrete cells, (b) a finite set of possible states associated to every cell, (c) a neighborhood of adjacent cells whose state influences the central cell, (d) uniform transition rules through time and space, and (e) a discrete time step to which the system is updated (Wolfram, 1984). A combination of Markov and cellular automata approaches has been shown to improve models describing complex natural patterns (Marshall and Randhir, 2008; Fan et al., 2008; Peterson et al., 2009).

The modeled landscapes were generated using the software IDRISI, the Taiga version. The MARKOV module of the software analyzes a pair of land cover images and outputs a transition probability matrix, a transition areas matrix, and a collection of conditional probability images. The transition probability matrix is a text file that records the probability of each land cover category changing to every other category. The transition areas matrix is a text file that records the number of pixels that are expected to change from each land cover type to each other land cover type over the specified

number of time units. The conditional probability images report the probability of each land cover type to be found at each pixel after the specified number of time units. These images are calculated as projections from the later of the two input land cover images. The CA_MARKOV module then utilizes the transition area matrix and a base-image to model future landscapes. In addition, the CA_MARKOV module integrates the spatial information based on the conditional probability images, where the CA model changes a particular pixel from one land cover class in time step i to another class in time step j based on the state of the local neighborhood. In other words, the CA model utilizes a collection of the conditional probability images (also known as the suitability images) to start an iterative process of relocating the pixels to the proximity of the same land cover class until it meets the area predicted by the Markov model for each land cover class.

For this study, I used land cover maps (spatial resolution: 30 m x 30 m) from 1977 and 1989 (Fig. 4-2a and 4-2b) as input maps in a CA-Markov model to predict changes for 2000 and 2007 (Fig. 4-2e and 4-2f) with the assumption that the pre-1989 drivers acting on the landscape are the only drivers guiding post-1989 changes. This assumption obviously was not met in reality as tree felling was banned within the park after 1989 as a result of changes in national-level policy and other conservation strategies were implemented in the surrounding areas during the 1990s. However, this assumption is essential in my current counterfactual approach reconstructing an alternative scenario with an absence of the national-level policy intervention.

Both of the transition probabilities for 2000 and 2007 were calculated by the MARKOV module based on the transitions between 1977 and 1989. The CA_MARKOV module then used these transitions to predict spatial patterns for 2000 and 2007 using

the 1989 image as the base-image. Then I calculated predicted land cover areas from the transition matrices generated for 2000 and 2007. I also created two-date change trajectories for forest transitions for comparison purposes. Since I was interested in forest transitions of the area (i.e. classes changing to or from forest class or stable forest class) only, I did not report the stable non-forest class in the change trajectories. Pixels estimated to convert from non-forest to forest were grouped as “reforestation” and those estimated to convert from forest to non-forest were grouped as “deforestation”. Pixels estimated to remain as forest in both dates (i.e. in 1989 and 2000 for 2000 transition probabilities and in 1989 and 2007 for 2007 transition probabilities) were grouped as “stable forest”. I used a χ^2 test to compare the actual changes with the predicted changes, with higher χ^2 values indicating higher levels of dissimilarity between compared land covers across time and space.

Since it was not possible to directly validate the counterfactual scenarios, a land cover map for the year 2007 was predicted based on the maps from 1977 and 2000 for the validation purpose. Choosing the input maps from 1977 (well before the intervention) and 2000 (well after the intervention) enabled me to disregard the deforestation trend in the 1980s, and also to capture the current reforestation trend in the area in the predictive model. I tested this modeled landscape for accuracy using the training samples collected during summer 2008. I believed that this alternate modeled landscape would better represent the “actual” land cover if the assumption that the 1988 policy change is important in changing land cover trends in the study area, is in fact true.

Fragmentation Indices

The long-term impacts of policy intervention on forest fragmentation, along with forest cover, are also critical indicators of a park's success (Nagendra et al., 2006; Hartter and Southworth, 2009). Identifying trends in landscape fragmentation over time can provide us with insights on the conservation strategies impacting land cover change. I wanted to examine if land cover maps generated for the counterfactual scenario can be used to infer fragmentation trends. I ran multiple instances of the CA_MARKOV module with different numbers of CA iterations to generate a range of possible fragmentation scenarios. I generated 8 different scenarios resulting from the following iteration numbers: 5, 7, 9, 10, 13, 15, 20, and 25. Then I used FRAGSTATS 3.3 to calculate a set of landscape metrics for each of these scenarios to describe the spatial patterns of the forest class. The following landscape metrics (see McGarigal and Marks, 1995 for detailed description) were selected based on their capability to describe the spatial pattern without being redundant (Haines-Young and Chopping, 1996).

- *Number of patches:* the total number of patches.
- *Edge density:* total edge per unit area for the class, in meters per hectare.
- *Mean patch area:* average area for the class, in hectares.
- *Mean shape index:* average complexity of patch shape compared to a square patch of identical area. For a single patch, the range varies from 1 when square to increasing numbers without limit with increasing complexity.
- *Mean nearest neighbor distance:* average nearest neighbor distance in meters.
- *Interspersion juxtaposition index:* measurement of the degree of interspersion of patches of a class with other categories. This index ranges between 0 and 100, with lower values for increasingly uneven distribution of patches of different classes.

The above patch- and class-level indices were used to compare descriptive metrics of forest cover pattern across the four observed dates. These metrics were also

used to compare actual patterns from the years 2000 and 2007 with those predicted for the same years. I obtained an estimated distribution of the predicted values for each of these indices for both 2000 and 2007, and then tested the hypothesis if the observed values fall within 95% confidence interval around the predicted mean values.

Results

Changes in Land Covers and Forest Transitions

Forest covers as seen in actual changes and predicted changes, both within the park and in the surrounding buffer, are notably different in amount and show opposite trends (Fig. 4-2 and Fig. 4-3). Predicted forest cover for 1977- 2007, based on the CA model, shows a deforestation trend for the park and an almost stable trend for the buffer (Fig. 4-3a). However, the actual changes show reforestation, both for the park and the buffer. Bare soil class shows exactly opposite trends. An increase in the soil cover was predicted for the park, while the buffer was predicted with almost no change. The actual changes show a declining trend for both (Fig. 4-3b). A χ^2 test (p=0.05) confirms statistically significant dissimilarity between actual and predicted changes across time and space.

Actual and predicted forest transitions based on two-date change trajectory images (1989- 2000 and 1989- 2007) are shown in Fig. 4-4. The model predicts approximately 20% more deforestation in 2000, only within the park. For 2007, the predicted deforestation is even higher, at 26%. Deforestation in the buffer was predicted to be 10% (2000) up to 16% (2007) higher than what actually occurred. Predicted reforestation within the park was approximately 4% lower than the actual reforestation, both in 2000 and 2007. Predicted reforestation in the buffer was approximately 3% higher for 2000 and 7% lower for 2007 than the actual reforestation. The modeled

landscape was predicted with 20% (2000) to 26% (2007) less stable forest within the park when compared to the actual landscape. The model also predicts 9% (2000) to 17% (2007) less stable forest within the buffer compared to the actual landscape.

Changes in Fragmentation Patterns

The overall descriptive changes in selected patch- and class-level metrics for the forest class differ considerably for actual and predicted scenario (Fig. 4-5 and Fig. 4-6). If I consider only the actual changes across four dates, the year 1989 shows the maximum fragmentation in terms of the landscape metrics, within the park in particular. Both for the park and the buffer, the number of forest patches was lowest in 1977, which reached to its highest in 1989, and is showing a declining trend afterwards. According to the patch area metric, forest patches were largest in 1977, and the smallest in 1989, both for the park and the buffer. Both the increasing number and decreasing size of the forest patches indicate forest habitat loss and increased fragmentation during 1977-1989. The graphs for interspersion-juxtaposition index for the park and the buffer also indicate that forest patches were distributed more evenly in 1977 with respect to the other land cover classes, while the distribution was most uneven in 1989 representing highly fragmented landscape. As a result of this fragmentation, mean distance between forest patches was lowest in 1989, especially in the park. Evidently, establishment of the park did not prevent the landscape from being more fragmented. Post-intervention landscapes, however, seem to recover having larger forest patches with more uniform shape, decreasing edge density, and more uniform distribution with respect to the other land cover classes.

Multiple predicted scenarios, as generated by running multiple CA_MARKOV modules, resulted in a range of predictions for each of the landscape metrics (Fig. 4-5

and Fig. 4-6). I tested whether the observed value for a particular index could have come from the corresponding estimated distribution (at 95% confidence interval).The tests reveal that most of the fragmentation indices derived from the predicted land cover maps differ significantly from the actual patterns from the years 2000 and 2007, the predicted scenarios having larger and more complex patches with greater inter-patch distance, with a few exceptions (Fig. 4-5 and Fig. 4-6). In other words, the post-1989 real landscape consists of smaller even-shaped patches those are closer to each other, while the simulated landscape consists of bigger patches with higher edge densities, which are more complex shape-wise and farther away from each other. The predicted landscapes, both for the park and the buffer, also indicate towards an uneven distribution of forest patches with respect to the other land covers.

Model Validation

The overall accuracy of the alternate predicted map for 2007, based on the transitions between 1977 and 2000 in an attempt to validate the methodology, is 80%. While the model performed quite well in predicting the overall trend, areas of mismatch between actual and predicted landscapes include areas with post-2000 forest regeneration (north-eastern and south-western corners of the buffer), and seasonally flooded areas surrounding the reservoir in the northern part of the park (Fig. 4-2). Since, the prediction is based on transitions between 1977 and 2000, the model failed to accurately predict areas with post-2000 forest regeneration. Also, the model predicted the areas surrounding the reservoir within the park as “bare soil”, based on the 2000 image which was collected well after monsoon season. These areas are classified as “water” in the actual 2007 image which was acquired right after the monsoon season, preventing the model having greater accuracy. Overall however, this level of accuracy

reflects well on the use of this modeling technique to evaluate conservation interventions in this landscape.

Discussion

India has a network of 660 protected areas that includes 99 national parks covering an area of 39,048 km^2 (1.19% of the country) (National Wildlife Database, 2009). This network was created to help conserve a significant part of the country's biodiversity. The basic approach of park management in India has been exclusionary based on the assumption that permanent human settlement within or in near vicinity of the park degrades the ecosystem through resource extraction, which has been supported by several case studies (Barve et al., 2005; Karanth et al., 2006; Davidar et al., 2007). However, in the present case study, the park suffered from deforestation even after it was declared a national park, despite the fact that only one village is located within its boundaries. It was also suggested that the reason behind this forest cover loss within the park was not related to resource extracting communities, but rather is due to the regular forest management activities conducted by the state government (Mondal and Southworth, 2010).

The counterfactual scenario, as simulated by the CA-Markov model, is notably different from the actual one, both in terms of individual land covers (Fig. 4-2 and Fig. 4-3) and forest transitions (Fig. 4-4). The modeled scenario predicts 20%- 26% more deforestation within the park itself than actually occurred. Clearly, if there had there been no intervention in 1988 in the form of the policy amendment, the park would probably have continued to suffer from extensive deforestation, challenging the very idea of national park creation for conservation. However, we must use these numbers with caution and rather focus on the pattern. The actual scenario, even in the absence

of the intervention, could have been something different than that predicted in this study. The rotational forests have a cycle of 30 years and it is likely that there could have been other spots of forest clearings, rather than aggregated cleared lands in the south-western corner of the park as predicted (Fig. 4-2d). I attempted to capture the variability in the model predictions by generating multiple counterfactual scenarios with different CA iteration numbers, which might have an effect on the spatial patterns of the modeled landscapes (Fig. 4-5 and Fig. 4-6).

As mentioned earlier, all types of Indian forests used to be managed under several “working circles” with a 30-year extraction cycle until the 1988 forest policy amendment (MoEF, 1988). The findings of this study points out that in the absence of this amendment the park would have suffered continued deforestation, contrary to the purpose of its establishment. It could therefore be suggested that the intervention was indeed successful in limiting deforestation. What is interesting to note here is the intervention I chose was the revised policy in 1988 that banned tree felling within any national park, rather than the establishment of the park itself in 1975. This corroborates my argument that mere establishment of a national park, without favorable national-level policies and proper law enforcement, might not be effective from the conservation point of view.

The surrounding buffer, despite being managed for commercial extraction shows an encouraging trend of reforestation. At a broader scale, this reforesting trend has been credited to several factors including degree of protection, commercial plantation, radical policy changes, and social forestry (Mather, 2007; Nagendra, 2009). Particularly, the implementation of various state- and World Bank-funded forestry projects in the

1990s has been suggested to make the reforesting trend possible in my study area (Mondal and Southworth, 2010). my findings suggest that in the absence of these forestry projects achieving the reforesting trend might not have been possible. Hence it is important to have favorable forest management practices surrounding a national park in order to conserve a greater ecosystem and its biodiversity, otherwise, increasing deforestation in the surroundings would result in an “islandized park” (Hartter and Southworth, 2009).

While the predicted land cover maps suggest a positive outcome of the intervention in terms of forest conservation, the landscape metrics suggest something quite contrary to this. The predicted land cover maps show larger forest patches when compared to the actual maps, except for the year 2007 (Fig. 5 and Fig. 6). This might be an artifact of this approach, since the CA model prediction is solely based on the input images and is a neighborhood based analysis. Since the forested area at the south-western boundary of the park in the 1977 image was selectively cut between 1977 and 1989 and was classified as bare soil in the 1989 image, the CA-Markov model has assigned maximum bare soil ‘pixels’ surrounding this area based on its higher probability. This resulted in similar land cover patches clumped together, which in turn resulted in larger patches. As already mentioned, the actual scenario could have been different, even in the absence of the intervention, and so the fragmentation pattern could have been different. I have tried to capture this variability by reporting an estimated distribution for each of the metrics resulting from different counterfactual scenarios (Fig. 4-5 and Fig. 4-6). Nevertheless, the positive influence of the intervention within the park is indirectly supported by the actual patterns (Fig. 4-5). The actual

patterns show increased landscape fragmentation till 1989, even after the establishment of the park that not only ceased after the policy enforcement in 1988 but also continued to recover with time. Similarly, the predicted patterns for the buffer (Fig. 4-6) suggest that without the state- and World Bank-funded afforestation programs it would have been difficult to achieve the current reforesting trend necessary to maintain the greater ecological system.

CA-Markov models are extensively used to identify and describe a myriad of processes including land cover change (Fan et al., 2008), rangeland dynamics (Li and Reynolds, 1997), species composition (Silvertown et al., 1992), forest succession (Alonso and Sole, 2000), and watershed response (Marshall and Randhir, 2008). Although a recent study has utilized CA-Markov models to identify the role of different management eras of the former Soviet Union in changing forest cover patterns (Peterson et al., 2009), my study is unique in utilizing CA-Markov models to generate a counterfactual scenario. Moreover, this simple approach is easy to use with remote sensing data and is capable of empirical evaluation of protected area management. This approach is particularly useful in areas where conventional methods of measuring protected area effectiveness (Andam et al., 2008; Gaveau et al., 2009) are difficult to adapt. Model validation suggests the predicted changes to be governed by the input maps, and in turn the policy implementation, rather than an artifact of the approach. Since the module used in this study requires two land cover maps as inputs, the method could potentially simulate counterfactual landscapes in any other scenario. However, future research on different ecological and social settings is needed to establish this as a robust method.

Conclusion

I proposed an empirical tool that integrates counterfactual approach and CA-Markov modeling to evaluate conservation intervention. I tested this method to understand landscape dynamics in a human-dominated tropical landscape using a case-study from central India. I explored the importance of national level policies in conservation efforts asking a fundamental question: what would have happened had there been no intervention? I utilized the proposed method to simulate alternate scenarios for 2000 and 2007 for the study area, assuming the absence of the 1988 forest policy intervention. Findings of this study suggest there would have been more deforestation, both within the park and in the surroundings, in absence of the intervention. This study also suggests that integrating CA-Markov models and counterfactual approach can be effective in evaluating conservation initiatives and law enforcement. This approach can be applied to any socio-ecological settings, and is particularly useful in developing countries where landscape dynamics is often governed by a group of social, ecological, economical and political factors.

Table 4-1. Forest conservation acts/policies of India from 1947-1990. The list has been restricted to acts/policies that have been identified by academics and park managers to govern landscape dynamics in the study area.

Forest Acts/Policies	Salient Features
National Forest Policy, 1952	An extension of the colonial British Policy that the claims of communities living in and around forests should not override national interests; in other words the policy justified destruction of forests for the construction of roads, building of irrigation and hydro-electricity projects.
	Discouraged communities (mainly tribal) living in and around forests from using the forests.
	Focused on obtaining more and more revenue from the forests.
Wildlife Protection Act, 1972	Provided increased degree of protection to wildlife and their habitat, as hunting and harvesting scheduled plant or animal species became banned.
	Helped increase protected areas (national park and wildlife sanctuary) throughout the country.
	Had different provisions for communities claiming land rights within sanctuaries, including a) exclusion of such areas from the protected area boundary, and b) compensation and relocation outside the protected area boundary, with the consent of the land holders. However, national parks were given more protection with almost no boundary alteration and more restrictions on entry in the park.
	Banned grazing and collecting forest products within national parks.
Forest Conservation Act, 1980	Conversion of government-owned forested land to non-forest areas, such as cultivation or other land uses was highly restricted. However, this provision excludes conversion necessary for conservation, development and management of forests and wildlife, such as establishment of check-posts, fire lines, wireless communications and construction of fencing, bridges and culverts, dams, waterholes, trench marks, boundary marks, pipelines etc.
	Had provisions for tree felling under a scientifically prepared management plan within national park or sanctuary for improvement of wildlife and its habitat.
	Suggested that removed forest produce should be used to meet the needs of the people living in or around the park/sanctuary and not for commercial purposes. However, in case of large scale timber/non-timber removal disposal through sales is allowed on approval of the Central Government.
Revised Forest Policy, 1988 – an amendment of 1980's Forest Conservation Act	Banned tree felling in any national park.
	Identified the tendency to consider forests as source of revenue to be the reason behind depleted forest resources.
	Identified need for afforestation programs.
	Suggested involvement of tribal communities in protection, regeneration and development of forests.

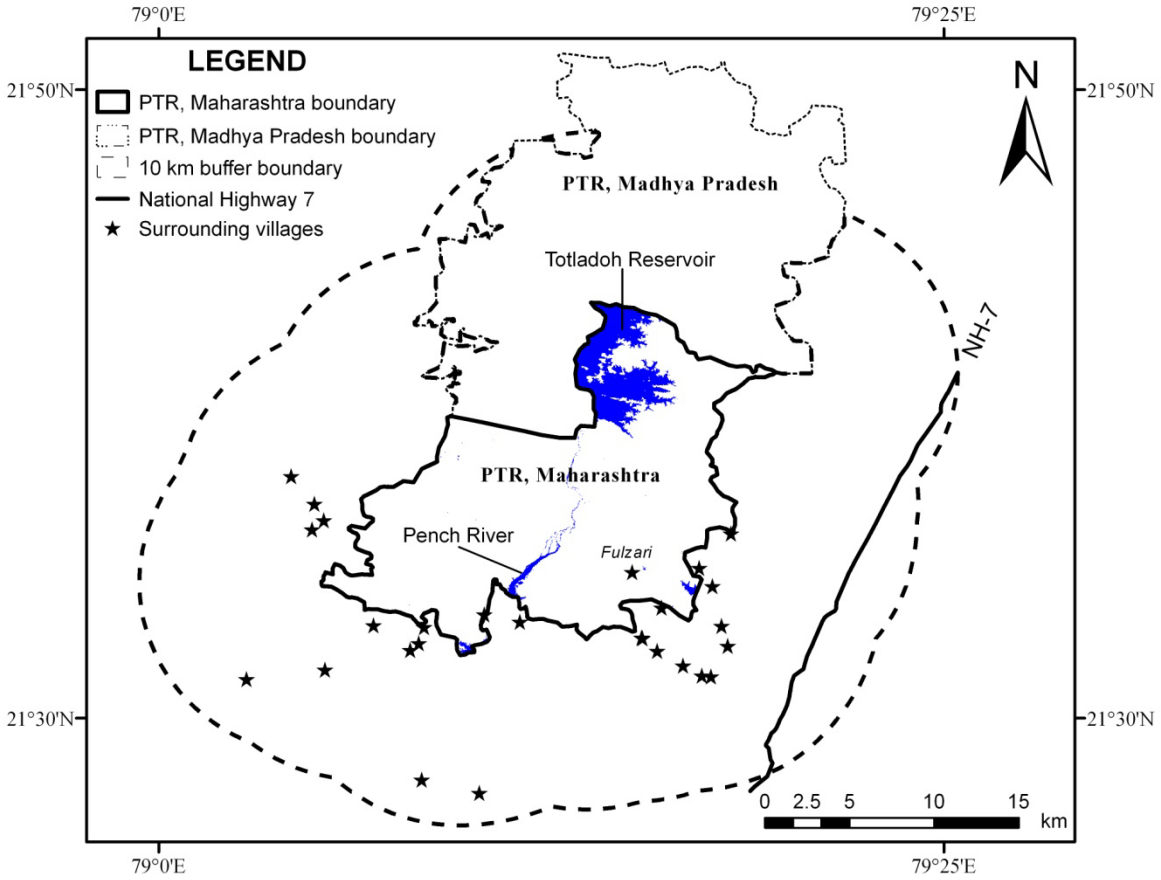


Figure 4-1. Map of the study area showing PTR and surrounding buffer up to 10 km, and visited villages in the area.

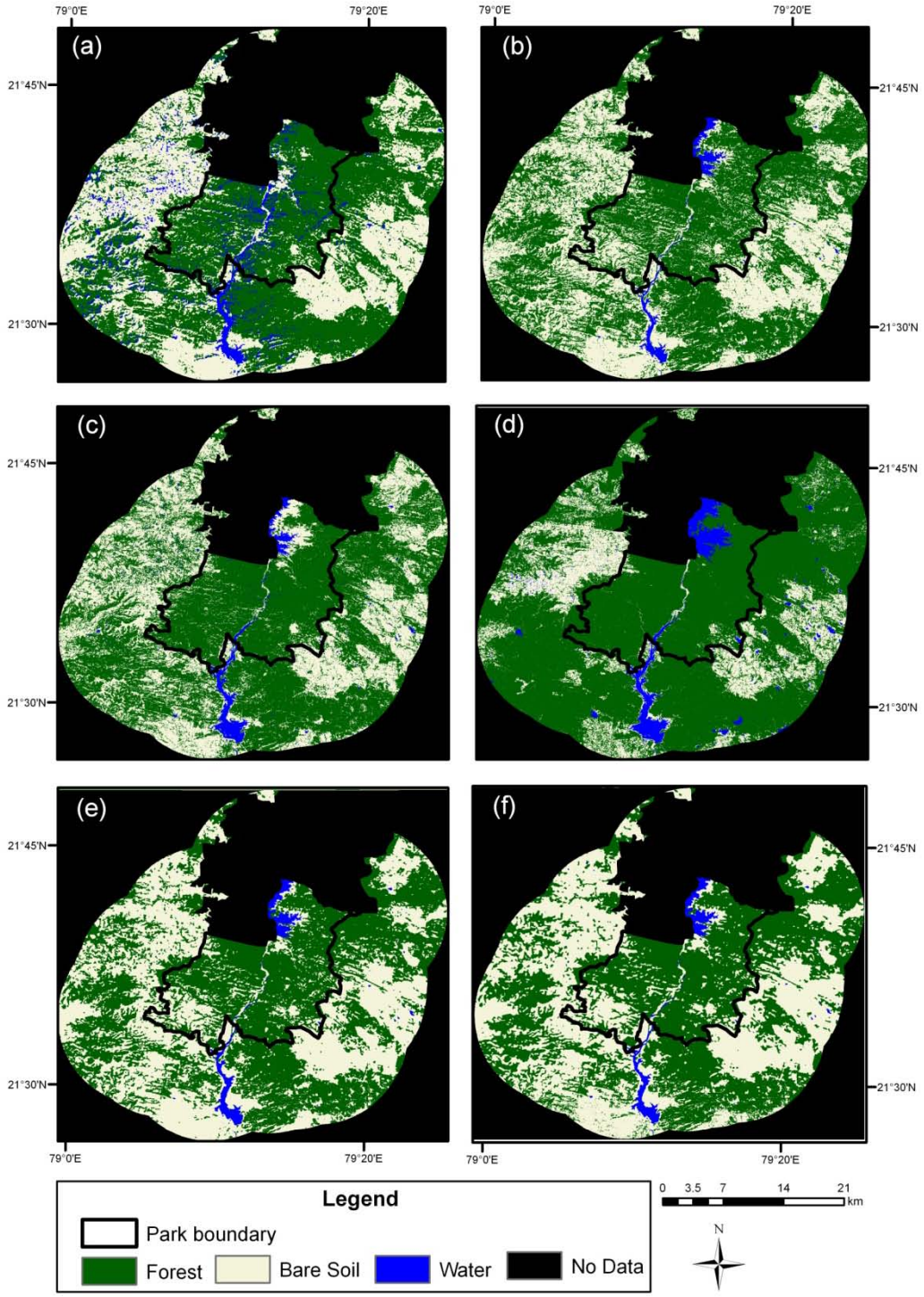


Figure 4-2. Land cover maps showing actual landscapes for (a) 1977, (b) 1989, (c) 2000, and (d) 2007, and CA-Markov model generated landscapes for (e) 2000 and (f) 2007 based on (a) and (b).

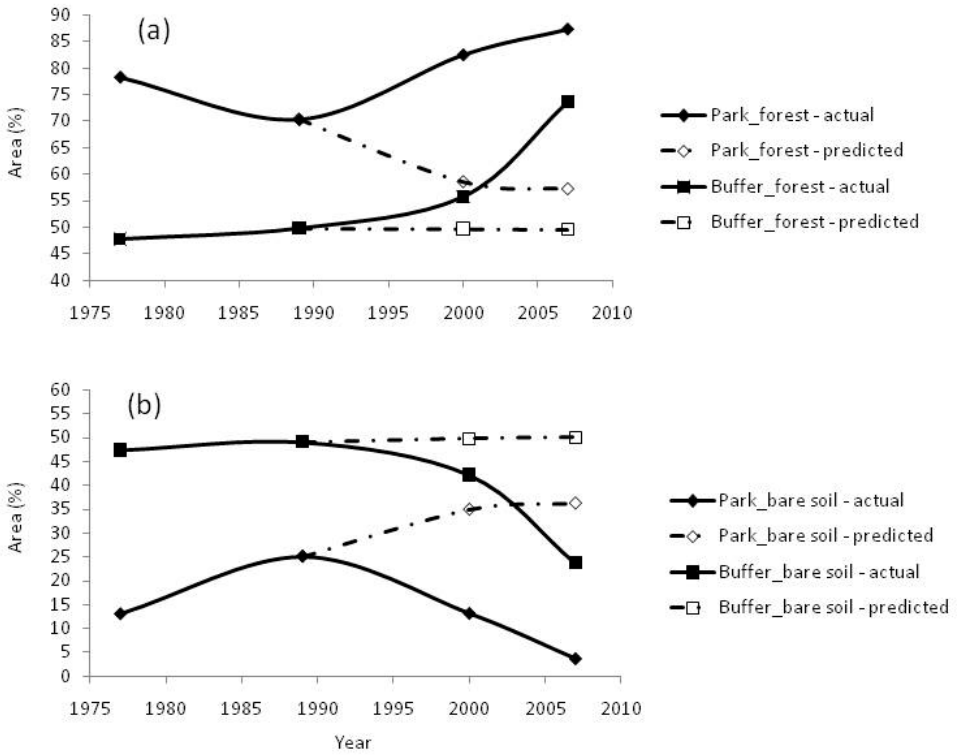


Figure 4-3. Actual and predicted land covers for the park and the buffer showing changes in (a) forest and (b) bare soil from 1977 to 2007.

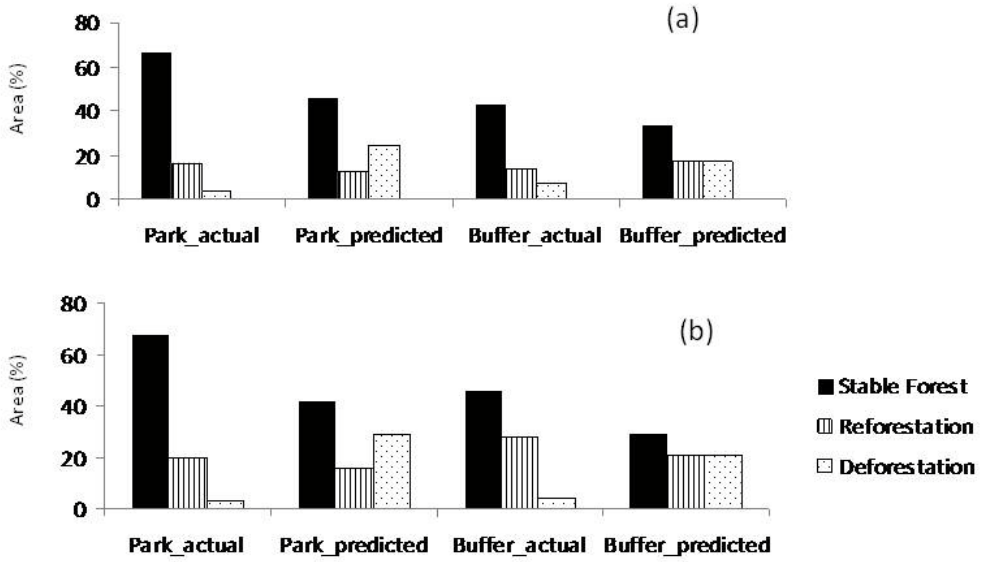


Figure 4-4. Two-date change trajectories for the park and the buffer showing actual vs. predicted forest transitions for (a) 1989–2000 and (b) 1989–2007.

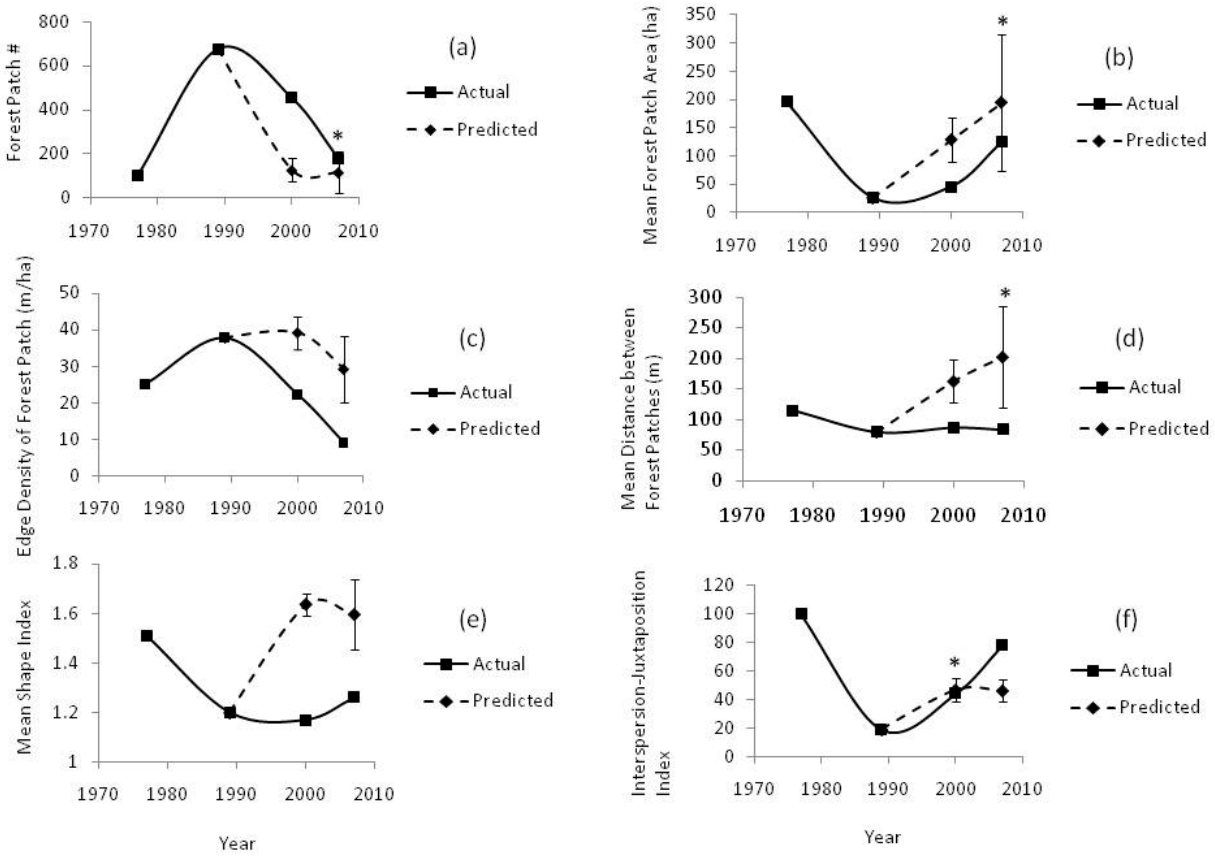


Figure 4-5. Series of landscape metrics for forest cover within the park. The graphs show actual and predicted changes in (a) patch number, (b) mean patch area, (c) edge density, (d) mean nearest neighbor distance, (e) mean shape index, and (f) interspersion-juxtaposition index. Multiple CA-Markov models with different numbers of CA-iterations were run to predict a range of forest fragmentation scenarios. Mean values are plotted along with 1 standard deviation for each of the predicted metrics. Actual values that fall within 95% confidence interval around the mean predicted values are marked by asterisks.

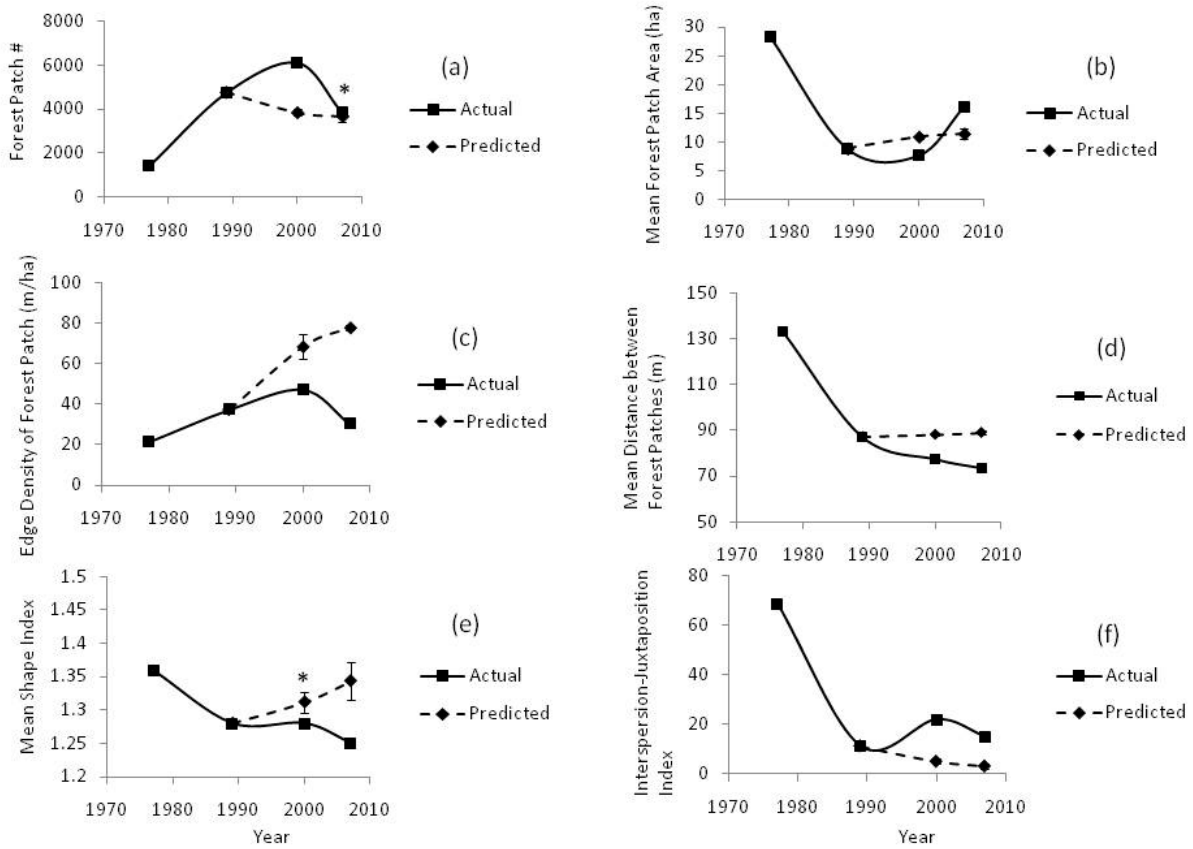


Figure 4-6. Series of landscape metrics for forest cover in the buffer. The graphs show actual and predicted changes in (a) patch number, (b) mean patch area, (c) edge density, (d) mean nearest neighbor distance, (e) mean shape index, and (f) interspersion-juxtaposition index. Multiple CA-Markov models with different numbers of CA iterations were run to predict a range of forest fragmentation scenarios. Mean values are plotted along with 1 standard deviation for each of the predicted metrics. For some of the metrics, the variability is smaller than the symbols. Actual values that fall within 95% confidence interval around the mean predicted values are marked by asterisks.

CHAPTER 5
QUANTIFYING SURFACE GRADIENTS WITH A 2-BAND ENHANCED VEGETATION INDEX (EVI2)

Introduction

Since its inception as a discipline, landscape ecology has maintained a strong focus in quantifying spatial and temporal heterogeneity, in order to link landscape patterns to organisms, communities or ecological processes (Turner et al., 2001). Ecological patterns could be an important key to identifying the processes that shape a landscape. Especially, in human dominated landscapes, ecological patterns could result from a complex interaction between several social, ecological, economic and political factors (Foster et al., 1998; Turner et al., 2001; Gardner and Engelhardt, 2008; Walsh et al., 2008), and hence can be used to identify underlying processes. With the patch-mosaic model as the operational paradigm, rapid advances have been made in quantitative landscape ecology in the past few decades. Most of these quantitative methods require a discrete patch-based representation of the landscape. While these methods have been quite useful in quantifying landscape patterns, error propagation through subjectivity and non-realistic assumptions of discrete landscape structure can not be completely ruled out (McGarigal and Cushman, 2005).

The patch mosaic model, also referred to as the patch-corridor-matrix model (Forman and Godron, 1986; Forman, 1995), views the surface as an aggregate of discrete patches and analyzes the surface properties through categorical representation of the landscape. The concept of the patch mosaic model is appealing to human intuition for its simplicity, consistency with well-developed and widely used quantitative techniques designed for discrete dataset and the availability of user-friendly software packages such as FRAGSTATS. While this model provides clear and concise end

products and has significantly contributed in understanding ecological pattern-process relationships (Turner, 2005), it is non-realistic in capturing surface heterogeneity and may result in a loss of important ecological information. Most surface processes are inherently continuous in nature, hence account for a rather ‘continuous’ representation of the surface properties. Another limitation of the patch-mosaic model is that it does not consider aspects of the third dimension, such as surface roughness, landform or relief variability (Hoechstetter et al., 2008) or intensity of the predictor variable (McGarigal and Cushman, 2005). While FRAGSTATS now offer a moving window analysis that generates continuous surfaces at different scales, which can be linked to organisms’ perspectives, the analysis is based on categorical maps and hence inherits all the limitations of the patch mosaic model. A conceptual shift has been proposed suggesting the surface gradient model as a supplement for the patch mosaic model to incorporate hierarchical and multi-scale ecological systems (McGarigal and Cushman, 2005). However, few studies test this new model in quantifying surface pattern.

In surface gradient model, it is possible to retain the inherent variability of a continuous predictor variable, without introducing subjectivity through defining sharp boundaries needed for categorization. Since the underlying heterogeneity of the predictor variable is preserved through space and across scale, it is possible to test a single predictor variable against multi-scalar response variables. Most importantly, continuous spatial and temporal patterns can be directly linked to ecological processes acting at different scales, without compromising organisms’ perspectives (Muller, 1998).

The surface spatial heterogeneity strongly depends on the predictor variable used to analyze it (Turner et al., 2001). Spatial and temporal remote sensing data have been

widely used to build ecosystem models about vegetation characteristics (Foody and Curran, 1994; Franklin, 1995; Kustas and Norman, 1996; Melesse et al., 2007; Stockli et al., 2008). The Enhanced Vegetation Index (EVI) is one such continuous variable that correlates well with several components of ecosystem dynamics such as leaf area index, biomass, canopy cover, and the fraction of absorbed photosynthetically active radiation (Boegh et al., 2002; Huete et al., 2006). EVI has been shown to be less prone to saturation in temperate and tropical forests making it more efficient over other widely used vegetation indices such as NDVI (Huete et al., 2006). Due to these properties EVI has been more effective for monitoring seasonal, inter-annual, and long-term variation in vegetation structure. EVI could be particularly useful in tropical regions where forest degradation or thinning is more prevalent than forest conversion and especially in places where discrete remote sensing classification techniques fail to capture subtle changes within a single class. Recently, a 2-band variation of EVI (EVI2) was developed for sensors with no blue band, such as MODIS and ASTER (Jiang et al., 2008; Stevens, 2009). EVI2 has been reported to correspond well with the original EVI (Jiang et al., 2008) and is expected to contribute towards a multi-sensor seamless vegetation index record. However, before using EVI2 to describe surface gradients, this variable needs to be tested for its capability to represent land cover diversity without loosing important information about the landscape.

To adopt the surface gradient models, landscape ecologists need to utilize a new suite of tools to analyze landscape structure. There are many promising techniques widely used in other disciplines, such as fractal analysis, spectral and wavelet analysis and surface metrology, which are theoretically suitable to analyze continuous variables

(McGarigal and Cushman, 2005; Cushman et al., 2010). Surface metrology, primarily a field developed for microscopy and molecular physics, is one such technique that holds promise for landscape surface analysis. In the past decade, several families of surface metrics have been developed and widely used in surface metrology. Scanning probe Image Processor (SPIP) software package has implemented such surface metrics that can quantify measures of surface amplitude in terms of its overall roughness, skewness and kurtosis, and total and relative amplitude (McGarigal et al., 2009). These techniques have made notable progress in the last decade, where they have been applied to different fields of study, such as in pharmaceutical science quantifying the intrinsic adhesion forces of lubricants and other pharmaceutical materials to a steel surface (Lee, 2004), in cell research quantifying the effect of certain hormone in kidney dysfunction (Oberleithner et al., 2004), and in polymer science mapping various surface properties of polymer-based materials (Munz et al., 2003). Theoretically these techniques hold promise for surface gradient approach in landscape ecology as well. Yet few studies employ these tools to analyze continuous surface gradients.

This research attempts to adopt the surface gradient model as the conceptual basis and applies some relatively less-known quantitative techniques for measuring surface gradients. This study explores EVI2 as an indicator of landscape heterogeneity as reflected in different land covers in a human-modified tropical landscape in Central India. Specific research questions addressed in this study are:

- Can EVI2 be used as a gradient surface to represent land cover diversity in a human-modified tropical landscape?
- How effective are the surface metrics, in terms of describing landscape composition and configuration?

Advantages and disadvantages of using such metrics are then discussed in the context of the current study.

Methods

Study Area

Four sample landscapes were delineated to capture the varying landscape composition and configuration (Fig. 5-1). These sample landscapes were chosen based on their land use history and different landscape compositions as shown by the EVI2 histograms (Fig. 5-2). Subsets 1 is located at the reserve boundary and is composed of protected forest, commercially managed forest and some agricultural lands. The only forest village in the reserve is located within subset 1. Subset 2 contains commercially managed forest and residential area with agricultural land. Subset 3 is located within the reserve and is almost entirely composed of protected forest. Subset 4 is composed of agricultural lands with sparse vegetation. Both the subsets 2 and 4 are located far from the reserve, and are different from each other regarding landscape composition and configuration.

Spatial Data

A mosaic of two ASTER images was used to calculate EVI2 for this study. Both of the images are from the same date to minimize variability and noise. Images were geometrically rectified, radiometrically calibrated, and atmospherically corrected (Adler-Golden et al., 1999; Abrams et al., 2002). EVI2 was calculated as: EVI2 = 2.5*[(NIR-RED)/(NIR+2.4*RED+1)] (Jiang et al., 2008; Stevens, 2009), where NIR and RED are estimated surface reflectance values for near-infrared and visible red bands (ASTER bands 3 and 2, respectively). EVI2, like NDVI, is a unitless continuous variable and ranges between -1 and 1. Remote sensing variables, other than EVI2, were also

extracted from the images in order to include them in the statistical analysis. These variables include estimated surface reflectance values derived from each of 9 ASTER non-thermal bands (Table 5-1), one temperature image with blackbody surface temperature values calculated from all five ASTER thermal bands, and NDVI values calculated from estimated surface reflectance values for ASTER bands 2 and 3. A discrete categorical map (accuracy 94%, see Chapter 3 for details) was included as a proxy for landscape heterogeneity to check the utility of EVI2 to represent land cover diversity.

The landscape shows distinct characteristics in terms of the remote sensing variables used in this study (Fig. 5-3). Discrete classification of this landscape identified 3 broad land cover classes: forest, bare soil and water (see Mondal and Southworth, 2010 for details). The land cover map suggests somewhat equal proportion of forest and non-forest (cleared forest, bare agricultural land, and water) components in the entire landscape. However, forest is the dominant land cover within the tiger reserve. The land cover map also suggests extensive agricultural land and cleared forest (identified as bare soil in the map) surrounding the reserve. Forested areas, within or outside the reserve, are associated with lower blackbody surface temperature, and higher EVI2 and NDVI values. Areas designated as bare soil in the land cover map, are readily distinguishable for their higher blackbody surface temperature values (Fig. 5-3). These are also the areas with lower EVI2 and NDVI values. Water bodies also have lower EVI2 and NDVI values; however have lower blackbody surface temperatures unlike bare soil.

Statistical Analysis

Prior to any statistical analysis all images were rescaled to 30 m x 30 m pixel size. A representative landscape was chosen as a subset from the image for statistical analyses to reduce computing time (Fig. 5-1). Then the variables were converted in an ASCII dataset to be imported in the statistical software package PASW Statistics 18. Since, this analysis included both categorical and continuous variables, and linearity among these variables was not out of question, eta (η) coefficient was used to measure the correlation. *Eta* is equal to Pearson's *r* for linear relationship, but is greater than *r* for non-linear relationship. Eta^2 describes the percent of variation in the dependent variable described linearly or non-linearly by the independent variable. A correlation matrix was calculated, which examines the level of interrelationships among the reflective bands, estimated surface temperature, vegetation indices and discrete land cover classes (Table 5-2). *Eta* varies from 0 to 1 and has no sign. Although the eta coefficient can not be used to establish causal direction, it is a useful coefficient to examine correlations among mixed variables of non-linear association (Southworth, 2004; Garson, 2008).

A discriminant function analysis was used to identify the relative capability of EVI2 to differentiate between different land covers. The Wilk's lambda coefficient was used to determine the relative importance of input variables in discriminating land covers (Table 5-3). The magnitude of the lambda coefficient (0.0–1.0) is inversely proportional to the ability of a variable to discriminate between the classes (Boyd et al., 1996).

Surface Metrics

This study utilized some of the metrics generated by the software SPIP and examined their utility in quantifying surface properties. SPIP generated metrics can be

of both spatial and non-spatial in nature. While non-spatial metrics can explain landscape composition, spatial metrics can be used to describe landscape configuration. Spatial metrics incorporate spatial information by describing variability in the overall distribution of the gradient surface as well as the spatial arrangement, location or distribution of peaks/valleys in the gradient surface. Non-spatial metrics are sensitive to overall height distribution in the gradient surface, but not the spatial arrangement, location or distribution of peaks/valleys in the gradient surface. In this study both spatial and non-spatial metrics were included. Specifically, the following metrics were chosen for their relevance to landscape ecology and to avoid redundancy (McGarigal et al., 2009). Selected metrics are described here briefly, for a detailed description please refer to McGarigal et al., 2009 and SPIP documentation (SPIP™).

- *Surface skewness (Ssk)*: describes the asymmetry of surface height distribution about the mean. A surface with dominantly higher values (or 'peaks') and occasional lower values would have negative *Ssk* values, while a surface with dominantly lower values (or 'valleys') and occasional higher values would have positive *Ssk* values. High skewness (positive or negative) denotes a landscape with dominant surface heights, i.e. extreme values in the surface properties. Landscapes with somewhat even distribution of peaks and valleys would have *Ssk* values near to zero. For this study, landscapes with dominant vegetation would have mostly higher EVI2 values with occasional lower values for non-vegetated areas, hence are expected to have negative *Ssk* values. On the contrary, landscapes with mostly non-forested areas would have lower EVI2 values with occasional 'peaks' for vegetation, and hence are expected to have positive *Ssk* values.

- *Surface kurtosis (Sku)*: describes the shape of the surface distribution. Like *Ssk*, *Sku* is sensitive to high 'peaks' or deep 'valleys'. A landscape with relatively even distribution of surface heights about the mean is said to be 'platykurtic', with *Sku* values less than 3. A landscape with areas well above or below the mean (dominantly high peaks or deep valleys) has *Sku* values greater than 3 and is called 'leptokurtic'. For example, landscapes with dominant land covers (forest or non-forest) are expected to be leptokurtic, while landscapes with somewhat even proportion of land cover components are expected to be platykurtic.

- *Dominant texture direction (Std)*: returns the angle of the dominating texture in the image as calculated from the Fourier spectrum. *Std* values range between 0 and 180 degrees. This metric is only meaningful when there is a dominant texture in the

landscape; otherwise it returns a value of zero (i.e. for flat landscapes). The distribution of *Std* is given by a graph called 'angular spectrum', which represents relative amplitude for different angles as calculated by summation of the amplitudes along M equiangularly separated radial lines. This spatial metric could be particularly useful to locate parallel 'ridges' (e.g. forest corridors) embedded within other land cover components (e.g. matrix of non-forest), where texture direction would be parallel to the direction of the ridges.

- *Texture direction index (Stdi)*: measures the relative dominance of the dominating texture direction and is calculated as the average amplitude sum divided by the amplitude sum of the dominating direction. *Stdi* values range between 0 and 1. Landscapes with dominant texture direction have values closer to 0, while landscapes with similar amplitude sum for all directions have a value equal to 1.

Ssk and *Sku* are called amplitude parameters and are non-spatial in nature. *Std* and *Stdi* are called spatial parameters as they provide spatial information about the landscape. The software requires the landscapes to be square in shape; hence all the four sample landscapes are 5 km x 5 km in extent.

Results

Statistical Analysis

EVI2 was identified as the most suitable continuous variable, among the ones used in this study, to represent the land cover diversity as suggested by the correlation and discriminant analysis results. EVI2 was highly correlated with land cover classes with η=0.85 (Table 5-2), only followed by NDVI with η=0.82 and temperature with η=0.74. The reflected bands have lower η values between 0.59 and 0.67. EVI2 also had the minimum value for the lambda coefficient (0.28), meaning that it has a high capability of differentiating between different land covers (Table 5-3). While both the vegetation indices (EVI2 and NDVI) performed quite well in terms of representing land cover diversity, EVI2 has an advantage over NDVI in tropical regions with high biomass, as in the present study area. However, the land cover map used in this study has very

low thematic resolution. Hence, correlation of EVI2 to land cover maps with higher thematic resolution should be further tested.

Surface Metrics

Ssk values are negative for all the landscapes except subset 4 (Table 5-4). Negatively skewed surface distribution, for this particular study, means dominance of high EVI2 values, which in turn means dominance of vegetation in the sample landscapes. However, the magnitude of *Ssk* values for these landscapes vary greatly. Subsets 1 and 2 have comparatively higher negative values denoting dominance of vegetation with some deep 'valleys', i.e. non-vegetated area such as bare agricultural land, water and forest villages. Subset 3 has a very low negative value denoting mostly vegetated area with very little non-vegetated area. Subset 4 has a low positive *Ssk* value, representing a landscape of dominantly non-vegetated area with some vegetation.

Sku values for these four landscapes also support the above findings. All the landscapes except subset 4 have high (>3) *Sku* values (Table 5-4). Among these four landscapes, subset 1 has the highest kurtosis, which can be linked to the location of a forest village represented by very low EVI2 values (Fig. 5-3). Subsets 2 and 3 have moderately high *Sku* values, suggesting non-uniform distribution of vegetated and non-vegetated components in landscape composition. In addition, if we consider *Sku* and *Ssk* values for subset 3 together, it informs not only about dominance of forest cover but also about the lack of dominant non-forest land cover (i.e. deep valley). Subset 4 has *Sku* value less than 3, representing a landscape with somewhat uniform distribution of vegetated and non-vegetated surface. This typically represents agricultural fields with surrounding trees.

Spatial metrics for the sample landscapes suggest lack of any dominant texture direction. Most of the *Stdi* values are near to 1 (Table 5-4), meaning almost even distribution of “peaks” and “valleys” in this landscape. Subsets 2 and 3 have moderate *Stdi* values (0.6 – 0.7) suggesting weak to moderate texture in directions given by corresponding *Std* values (Table 5-4). Subsets 1 and 4 lack any dominant texture direction, as suggested by *Stdi* values (Table 5-4).

Discussion

Remote sensing variables have frequently been identified to be particularly suitable for monitoring dynamic environmental processes, such as the exchange of mass and energy among soil, vegetation, and atmosphere (Sellers et al., 1997; Mecikalski et al., 1999; Turner et al., 2003; Leuning et al., 2005). Monitoring rapid environmental changes, especially vegetation, under differing natural and anthropogenic influence has become increasingly efficient in the past few decades due to the availability of consistent remote sensing datasets. Vegetation indices, especially NDVI, are datasets which can be directly linked to the continuous surface processes, and are more likely to reflect the actual landscape with their inherent gradients. This study suggests that EVI2 is another important variable that can be used for environmental monitoring as a proxy for land cover diversity (Tables 5-2 and 5-3). Moreover, EVI2 has the capability to capture subtle changes in vegetation condition and structure, especially to differentiate between surface greenness and leaf area index (LAI) for vegetation with different background soil reflectance (Rocha and Shaver, 2009), and thus can be directly linked to land cover modifications. Designed for sensors without a blue band, EVI2 can not only be used for moderate to large-scale studies involving ASTER, MODIS and AVHRR datasets, but can also be utilized in multi-sensor

time-series analysis. The findings showed strong correlation between EVI2 and discrete land cover classes, suggesting the robustness of this variable as a representative for land cover diversity. Correlation of EVI2 and NDVI enabled confident use of EVI2 in this study, as NDVI has been previously used in many studies to successfully describe spatial heterogeneity (Kawabata et al., 2001; Garrigues et al., 2006; Gillespie et al., 2006; Garrigues et al., 2008).

Given their varied composition and configuration, the sample landscapes were expected to have varied surface metrics. Apparently, only non-spatial metrics performed well in bringing out the different characteristics of these landscapes (Table 5-4). Both *Ssk* and *Sku* values for subsets 1 and 2 indicate dominant surface heights for the landscapes, which can be interpreted as landscape dominance or evenness, analogous to the patch-based evenness metrics (McGarigal et al., 2009). The tiger reserve in the study area has maintained its forest cover in the last decade after showing some initial deforestation trend in the early 1980s (Mondal and Southworth, 2010). Hence, a uniform EVI2 distribution was expected for subset 3 which is a part of the reserve. The surface metrics not only identified dominance of vegetation in this subset, but also gave an idea of the relative proportion of vegetation and non-vegetation in the subset through its negative skewness and moderate kurtosis values (Table 5-4). The non-spatial metrics also accurately identified dominance of bare agricultural land (with low EVI2 values) with substantial presence of vegetation (‘peaks’ in EVI2 values) in subset 4, as reflected in its positive skewness and low kurtosis values (Table 4).

Spatial metrics, on the other hand, did not provide with any additional ecological information. According to the spatial metrics the sample landscapes lack any dominant

texture (Table 5-4). This is partly expected as subsets 3 and 4 visually lack strong texture orientation, being mostly dominated by vegetation and agricultural land respectively (Fig. 5-3). However, texture direction index failed to pick up any signal for sudden drop in EVI2 value due to the location of a forest village within the reserve in subset 1, returning a high *Stdi* value of 0.82 (Table 5-4). *Stdi* value of 0.60 for subset 2 somewhat supports the sharp boundary between forest and agricultural lands (Fig. 5-3). However, quite contrary to the expectation, subset 3 which is mainly composed of protected forest has moderate texture direction index (*Stdi*=0.70). Further examination of the landscape revealed the presence of fine scale linear non-vegetated elements (low EVI2), identified as the tributaries to the Pench River. It was interesting to note that this metric could identify such fine scale elements in a landscape, yet was not very sensitive to sharp boundaries between different land covers. This discrepancy could be explained by any of the following: a) the landscapes are visually misleading and actually lack strong texture or b) the metrics are not designed to capture texture orientation defined by sharp boundary between two land covers or sudden decrease in continuous variable at a landscape level. Also the texture orientation metric could be particularly sensitive to linear elements and only pick up signals when landscapes have such features. Whichever is the case, there is a strong need to test these metrics in other landscapes to evaluate their compatibility in landscape studies.

While some of the surface metrics used in this study were able to describe the landscape characteristics, there are definitely some questions about applicability of such metrics in landscape-level studies and robustness of these techniques. Even though there is a plethora of metrics available in SPIP, this study includes only four

metrics for specific reasons. Most of the SPIP generated metrics are sensitive to the unit of the predictor variable, thus cannot be used with unitless variables. Many of the SPIP generated metrics have theoretical applicability in landscape-level studies, such as summit density (*Sds*), surface area ratio (*Sdr*), root mean square slope (*Sdq*), dominant radial wavelength (*Srw*) (refer to SPIP documentation for detailed description of these metrics). However, SPIP did not generate any interpretable results for these metrics for the present study. This can be attributed to the fact that the software used in this study was developed for microscopy and molecular physics, which deal with very different spatial scales than landscape ecology studies. While theoretically these metrics hold promise to be efficient in ecological studies, the scale compatibility issue further underscores the serious need to develop user-friendly and easily available software compatible for landscape studies.

Conclusion

This study utilized the surface gradient model as a conceptual basis to quantify EVI2 as a surface gradient. This study explored the utility of some relatively less-studied tools in a tropical landscape. EVI2, used as a proxy for land cover diversity in this study, correlated well with the low resolution thematic land cover map. This study also suggests that there is potential for the surface metrics in ecological studies, provided availability of user-friendly tools and utilization of effective predictor variable. Currently, there are several available tools that can handle continuous variables to generate surface metrics. However, the challenge would be to tailor those tools for landscape ecology and/or develop new tools meant for landscape level studies.

Table 5-1. Characteristics of ASTER visible near-infrared (VNIR), short-wave infrared (SWIR) and thermal infrared (TIR) bands.

	Band	Spectral Resolution (µm)	Spatial Resolution (m)
VNIR	1	0.520 – 0.600	15 x 15
	2	0.630 – 0.690	
	3	0.760 – 0.860	
SWIR	4	1.600 – 1.700	30 x 30
	5	2.145 – 2.185	
	6	2.185 – 2.225	
	7	2.235 – 2.285	
	8	2.295 – 2.365	
	9	2.360 – 2.430	
TIR	10	8.125 – 8.475	90 x 90
	11	8.475 – 8.825	
	12	8.925 – 9.275	
	13	10.250 – 10.950	
	14	10.950 – 11.650	

Table 5-2. Correlation matrix showing the level of interrelationships among the reflected bands (Band 1 through 9), one blackbody surface temperature image (Temp) calculated from five thermal bands, vegetation indices (EVI2 and NDVI) and discrete land cover classes. The table lists eta coefficients (η) as a measure of non-linear association.

	Band1	Band2	Band3	Band4	Band5	Band6	Band7	Band8	Band9	Temp	EVI2	NDVI	Class
Band1	1.00												
Band2	.97	1.00											
Band3	.72	.69	1.00										
Band4	.88	.88	.87	1.00									
Band5	.89	.91	.82	.97	1.00								
Band6	.90	.91	.83	.98	.99	1.00							
Band7	.89	.90	.82	.97	.99	.99	1.00						
Band8	.90	.92	.82	.97	.98	.99	.99	1.00					
Band9	.87	.90	.80	.95	.98	.98	.98	.98	1.00				
Temp	.69	.73	.39	.60	.73	.72	.74	.75	.74	1.00			
EVI2	.81	.86	.88	.83	.81	.82	.82	.82	.81	.86	1.00		
NDVI	.84	.87	.84	.85	.85	.86	.86	.86	.85	.93	.95	1.00	
Class	.62	.67	.59	.62	.62	.62	.63	.63	.61	.74	.85	.82	1.00

Table 5-3. Discriminant analysis results showing the relative capability of input variables in differentiating between different land cover classes. The magnitude of the Wilks' lambda coefficient is inversely proportional to the ability of a variable to discriminate between the classes and is sorted in ascending order in the table.

Variables	Wilks' Lambda
EVI2	.28
NDVI	.33
Temp	.45
Band2	.55
Band4	.61
Band5	.61
Band6	.61
Band7	.61
Band8	.61
Band1	.62
Band9	.62
Band3	.65

Table 5-4. Surface metrics calculated from EVI2 for four sample landscapes from the study area. *Ssk* and *Sku* are non-spatial metrics denoting landscape composition, while *Std* and *Stdi* are spatial metrics describing landscape configuration. All of these are unitless metrics, except *Std* which is measured in degrees.

Surface Metrics	Subset 1	Subset 2	Subset 3	Subset 4
Surface Skewness (Ssk)	-1.73	-1.61	-0.25	0.33
Surface Kurtosis (Sku)	8.44	5.24	4.37	2.84
Dominant Texture Direction (Std)	74.71	89.53	78.99	83.81
Texture Direction Index (Stdi)	0.82	0.60	0.70	0.88

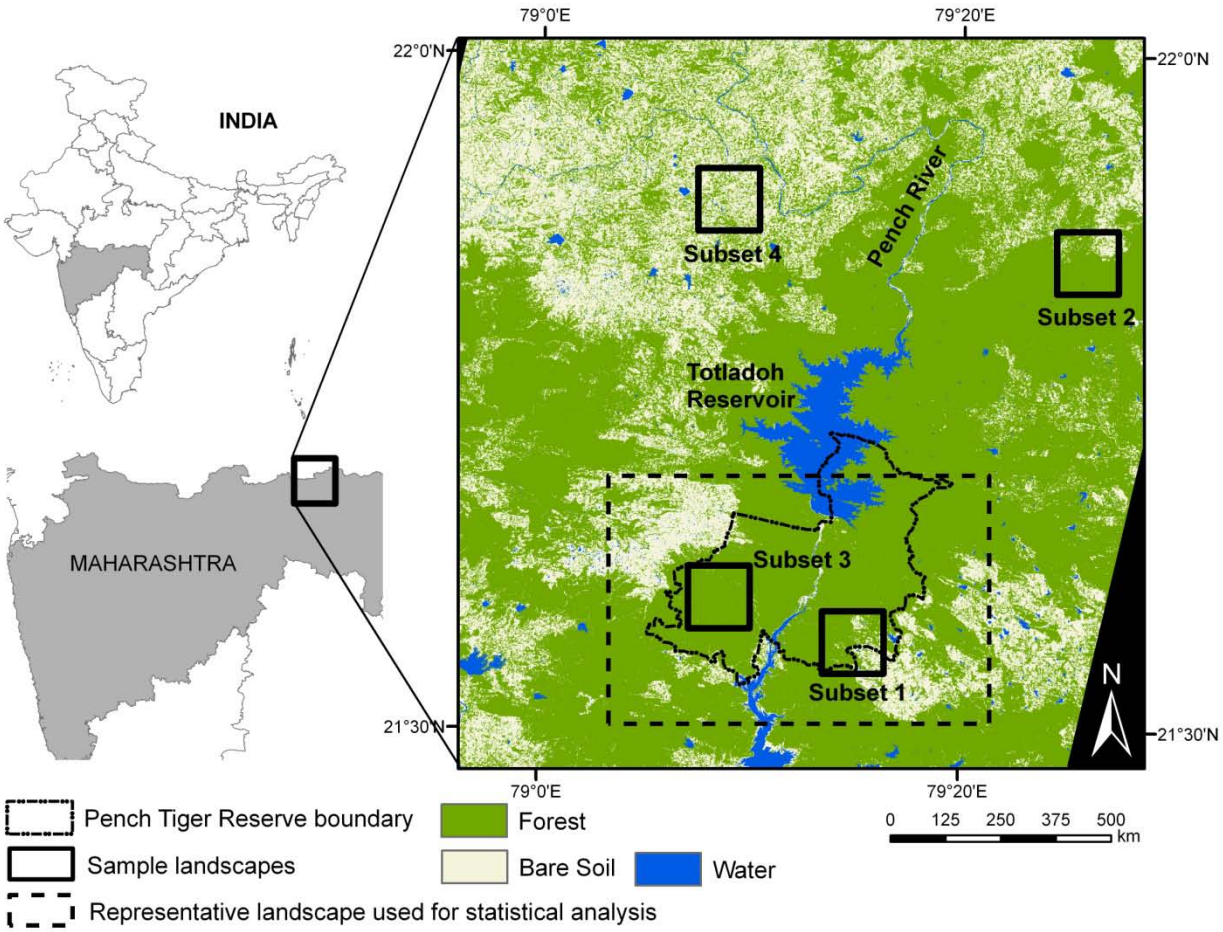
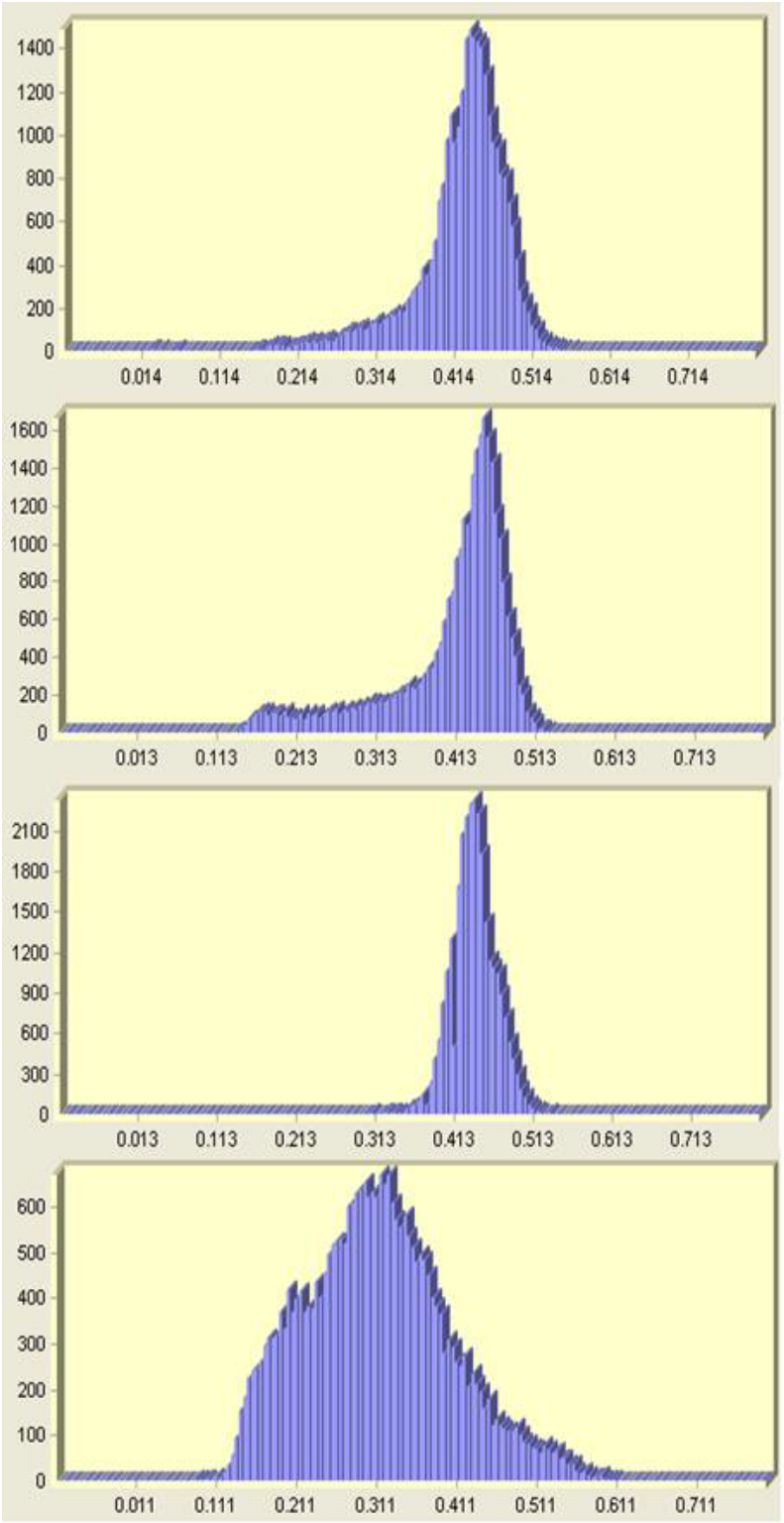


Figure 5-1. Land cover map shows locations of four sample landscapes used in this study in reference to the Pench Tiger Reserve.



(a) Subset 1

Minimum= -0.046
Maximum= 0.652
Mean= 0.424
Standard deviation= 0.06

(b) Subset 2

Minimum= -0.042
Maximum= 0.644
Mean= 0.408
Standard deviation= 0.077

(c) Subset 3

Minimum= 0.233
Maximum= 0.556
Mean= 0.439
Standard deviation= 0.029

(d) Subset 4

Minimum= 0.071
Maximum= 0.662
Mean= 0.311
Standard deviation= 0.091

Figure 5-2. EVI2 histograms for four sample landscapes.

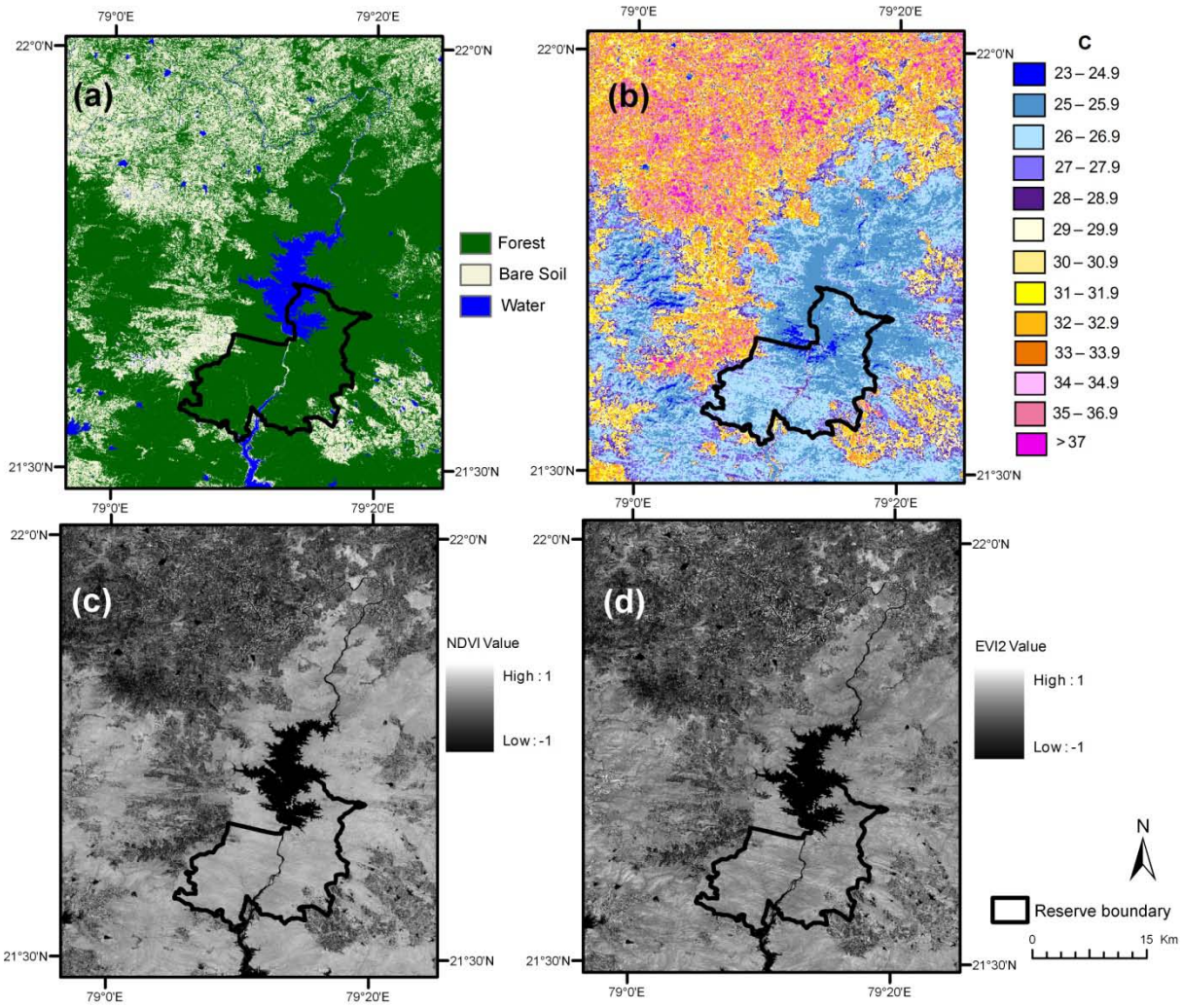


Figure 5-3. Spatial patterns in the study area as demonstrated by (a) land cover map, (b) blackbody surface temperature (˚C) image, (c) NDVI image, and (d) EVI2 image.

CHAPTER 6
COMPARISON OF ASTER-DERIVED THERMAL INDICES AND TASSELED CAP NORMALIZED INDICES IN DETECTING TROPICAL FOREST SUCCESSIONS

Introduction

Successional forests differ in their average height and basal area, the stand volume and basal density, and physiognomic characteristics (Moran and Brondizio, 1998). Quantification of specific forest parameters is important to better understand the impact of different successional stages on ecosystem changes, specifically water and nutrient cycling and carbon sequestration (Lu et al., 2003; Song et al., 2007). The first step to this quantification would need detailed mapping of different successional stages. Moderate resolution satellite images, such as those from Landsat, have been successfully used for such mapping purposes. However, technical issues related to longevity and quality of Landsat sensors warrant for assessment of alternate moderate resolution satellite images for such mapping purposes. While ASTER has comparable spatial, spectral and temporal resolutions making it effective replacement for Landsat, few studies report the capability of ASTER-derived vegetation indices or thermal indices in detecting tropical forest successions.

The tasseled cap indices, primarily developed for Landsat sensors, have been widely used for mapping and monitoring land cover changes, especially subtle changes in vegetation condition (Cohen and Spies, 1992; Cohen et al, 1995). These indices, when normalized, often highlight a certain land cover characteristics (Healey et al., 2005). For example, brightness, greenness, and wetness indices normalized for a “dense forest” class would record spectral distance of a given pixel from the dense forest centroid of the corresponding indices for a particular scene (Masek et al., 2008), thus highlighting disturbed forest pixels. These indices require linear combinations of

individual channels specific to each sensor. While coefficients for tasseled cap transformations have been proposed (Yarbrough et al., 2005) for ASTER, the utility of ASTER-derived tasseled cap indices in identifying vegetation structure or composition is yet to be tested.

Use of thermal infrared emissivities as an indicator of land degradation is a fairly new approach within remote sensing realm and is advantageous in many ways. Emissivity is defined as the ratio between actual emitted radiation and emitted radiation from a blackbody at the same temperature. Emissivity is independent of temperature itself and varies spectrally according to surface composition and geometry. These latter properties make emissivity observations potentially useful for land cover characterization. Along with emissivities, surface temperature also holds promise to effectively detect forest successions.

This study compares ASTER-derived tasseled cap indices, surface temperature and emissivitity in detecting tropical forest successions. Specifically, this study seeks to identify the degree of association of the remote sensing indices to different forest successions derived from historical land cover maps.

Methods

Study Area

This study utilized a heterogeneous landscape located in Central India as a template for tropical ecosystem (Fig. 6-1). The study area is dominated by tropical deciduous forest, especially teak. Some of these forests are protected as the landscape hosts Pench Tiger Reserve. The rest of the forests are commercially managed and supports subsistence livelihood of the local people. Interplay of changing forestry policies, various degrees of protectionism and local livelihood has resulted in

successional forests in this landscape, which is representative of many developing countries.

Data Pre-processing

Two scenes of ASTER L1-B data, acquired on October 30, 2007, were downloaded from LPDAAC for path 144/row 45. These images were generated by applying coefficients for radiometric calibration and geometric rectification (Abrams et al., 2002). Because ASTER images come with three spatial resolutions, visible infrared bands (bands 1 through 3) and shortwave infrared bands (bands 4 through 9) were fused to 30m pixel resolution. To reduce between-scenes variability, the at-sensor reflectance was calculated based on the following equation:

$$\rho_p = \pi \cdot d^2 \cdot L_\chi \cdot (E_{sun,\chi} \cdot \cos(\theta_{sz}))^{-1} \quad (6\text{-}1)$$

where d is the astronomical distance of the earth to the sun, L_χ is the at-sensor radiance, θ_{sz} is the solar zenith angle in radians, and $E_{sun,\chi}$ is the solar spectral irradiance as calculated by World Radiation Center (Yarbrough et al., 2005). The two at-sensor reflectance images were then mosaicked and later used to calculate tasseled cap indices.

Thermal atmospheric correction is a necessary step to approximate and remove atmospheric contributions from thermal infrared radiance data. Thermal bands (bands 10 through 14) were atmospherically corrected using the in-built function of ENVI 4.5 software package, which used the In-Scene Atmospheric Compensation (ISAC) algorithm (Johnson and Young, 1998). Assumptions of the algorithm include uniformity of atmosphere over the data scene, presence of a near-blackbody surface within the scene and absence of reflected downwelling radiance. Upwelling atmospheric radiance and atmospheric transmission are approximated using the Planck function and

assumed emissivity of 1. The two atmospherically corrected scenes were then mosaicked to cover the study area.

Successional Forest Subsets

I extracted two forest subsets for this study, for which I used a change detection map generated for a previous study (Mondal and Southworth, 2010) (Fig. 6-1). The first subset is for mature forests that have been stable for last 30 years (1977 onwards, just for park). For this group, I included areas those were forests on all four dates (1977, 1989, 2000 and 2007) in the four-date change detection map. However, forested areas on these four dates do not necessarily represent mature forest, as it is possible for these areas to witness land cover changes between any two years. To overcome this limitation, I extracted data only for the park, hopefully increasing the probability of the stable forest subset truly representing mature forest because of the protected status of the park since 1975. The other subset is for regenerated forests with age less than 7 years. For this group I included areas those converted from any non-forest land cover in 2000 to forest in 2007.

Emissivity and Surface Temperature Extraction

This study used emissivity normalization method to separate emissivity and temperature values from atmospherically corrected thermal infrared radiance data. This technique calculated the temperature for every pixel and band in the data using a fixed emissivity value (1 for this study). The highest temperature for each pixel is then used to calculate the emissivity values using the Planck function. While multiple methods for separating temperature and emissivity data exist, emissivity normalization method has been reported to be slightly superior to the others in a heterogeneous landscape dominated by gray body targets, such as vegetation (French et al., 2008). The fixed

emissivity value for this study was chosen as 1 to minimize the correlation with any particular land cover, as I was interested to compare the emissivity values derived from all of the 5 ASTER thermal bands in identifying different forest successions. Besides, at the 90 m spatial resolution of the thermal bands, vegetation is viewed as a canopy and multi-scattering effects would increase maximum emissivity to over 0.99 (Salisbusy and D'Aria, 1992). In addition to the emissivity values for all the 5 thermal bands, 3 new emissivity indices were developed using combinations of emissivity values for bands 10, 11 and 12. The formulas for these indices are listed under Table 6-1.

Normalized Tasseled Cap Indices

The at-sensor reflectance values for visible infrared and shortwave infrared bands were utilized for tasseled cap transformation using the transform coefficients developed for ASTER at-sensor reflectance (Yarbrough et al., 2005). The three principal components in the direction of brightness, greenness, and wetness were further used to develop normalized tasseled cap indices as described below. I used the stable forest mask to generate the normalized tasseled cap indices, such that (for example):

$$B' = (B - \mu_B) / \sigma_B \qquad (6\text{-}2)$$

where B is the tasseled cap brightness index for any pixel, μ_B is the mean tasseled cap brightness index for the stable forest class, and σ_B is the standard deviation of brightness within the stable forest class.

Statistical Analysis

The two subsets were used to extract data for each of the vegetation and thermal indices used in this study. A smaller data set was chosen for statistical analysis through random sampling. Then the variables were converted in an ASCII dataset to be imported in the statistical software package PASW Statistics 18. As a measure of

correlation eta (η) coefficient was used, since, unlike Pearson's *r*, eta can be used for both linear and non-linear relationships. Eta^2 describes the percent of variation in the dependent variable described linearly or non-linearly by the independent variable. *Eta* varies from 0 to 1 and has no sign. Although the eta coefficient can not be used to establish causal direction, it is a useful coefficient to examine correlations among mixed variables of non-linear association (Southworth, 2004; Garson, 2008). (Table 6-2).

A discriminant function analysis was used to identify the relative capability of the thermal and vegetation indices included in this study to detect the forest successions. The Wilk's lambda coefficient was used to determine the relative importance of input variables in discriminating two forest classes (Table 6-3). The magnitude of the lambda coefficient (0.0–1.0) is inversely proportional to the ability of a variable to discriminate between the classes (Boyd et al., 1996).

Results

Eta coefficient suggests strongest correlation between surface temperature and forest successions closely followed by two of the normalized emissivity indices (Table 6-2). Individual emissivities for bands 10, 11, and 12 performed better than the normalized tasseled cap indices and emissivities for bands 13 and 14. Discriminant function analysis suggests similar results with minimum Wilk's lambda coefficient for surface temperature, followed by two of the new emissivity indices (12by10 and 11by10). Normalized greenness and wetness indices were least capable to detect forest successions (Table 6-3).

Discussion

Landsat tasseled cap indices have been used to detect successional stages in various forest ecosystems, such as conifer forest (Cohen and Spies, 1992), hardwood

and softwood forests (Jin and Sader, 2005), and tropical forests (Lu et al., 2004). Normalization of these indices is expected to highlight deviation of spectral signatures of any given pixel from that of a stable forest. However, for the present study normalized greenness and wetness indices failed to correlate with forest masks with different ages.

Unlike vegetation indices, emissivity responds to plant canopy geometry and patterns between plant canopies. Hence it was expected to detect any successional changes in the present landscape. Previous study has reported capability of ASTER band 11 emissivity to detect degradation in sparsely vegetated terrain (French et al., 2008). This was possible as high emissivity vegetation is clearly distinguished from low-emissivity soil in a moderate resolution image. However, in a vegetation dominated region, such as my study area, it is difficult to distinguish between different successional stages by emissivity alone because of the very small emissivity ranges. The new emissivity indices slightly improved the performance, as they tend to highlight the small differences between emissivity values for different bands and different vegetation structure.

The land surface temperature performed best among the used variables to detect successional changes. Surface temperature reflects canopy temperature, rather than near-surface temperature, in tropical forests. Hence mature and old forests often have a lower temperature than young open-canopy forests (Southworth, 2004). In this study, however, some areas classified as the stable forest have higher temperature (Fig. 6-2). These areas most probably do not represent mature forest, but are classified as forest in all four dates resulting in being classified as stable forest in the four date change trajectory image. To address this issue I included stable forest only from the park.

This study highlights the need of further research to investigate the capability of ASTER-derived vegetation and thermal indices in detecting forest composition, structure, and density. Landsat-derived indices have been widely used in mapping and monitoring forest characteristics. Applicability of ASTER-derived indices in such purposes would ensure long-term record of forest characteristics.

Table 6-1. Calculation for newly developed emissivity indices. The numbers 10, 11 and 12 represent emissivity values for ASTER bands 10, 11, and 12 respectively

New Emissivity Index	Calculation
11by10	11 – 10/11 + 10
12by10	12 – 10/12 + 10
11by12	11 – 12/11 + 12

Table 6-2. The level of interrelationships of forest successions derived from change detection map with remotely sensed thermal and vegetation indices. The table lists eta coefficients (η) as a measure of non-linear association and is sorted in ascending order in the table.

Remote sensing variables	Eta	Eta^2
Surface temperature	0.566	0.321
Index 12by10	0.536	0.288
Index 11by10	0.525	0.276
Band 12 emissivity	0.470	0.221
Band 10 emissivity	0.456	0.208
Band 11 emissivity	0.376	0.141
Index 11by12	0.318	0.101
Normalized tasseled cap brightness	0.296	0.088
Band 14 emissivity	0.094	0.009
Band 13 emissivity	0.058	0.003
Normalized tasseled cap greenness	0.014	0.000
Normalized tasseled cap wetness	0.004	0.000

Table 6-3. Discriminant analysis results showing the relative capability of input variables in detecting tropical forest successions. The magnitude of the Wilks' lambda coefficient is inversely proportional to the ability of a variable to discriminate between the classes and is sorted in ascending order in the table.

Remote sensing variables	Wilk's lambda
Surface temperature	0.679
Index 12by10	0.712
Index 11by10	0.724
Band 12 emissivity	0.779
Band 10 emissivity	0.792
Band 11 emissivity	0.859
Index 11by12	0.899
Normalized tasseled cap brightness	0.912
Band 14 emissivity	0.991
Band 13 emissivity	0.997
Normalized tasseled cap greenness	1.000
Normalized tasseled cap wetness	1.000

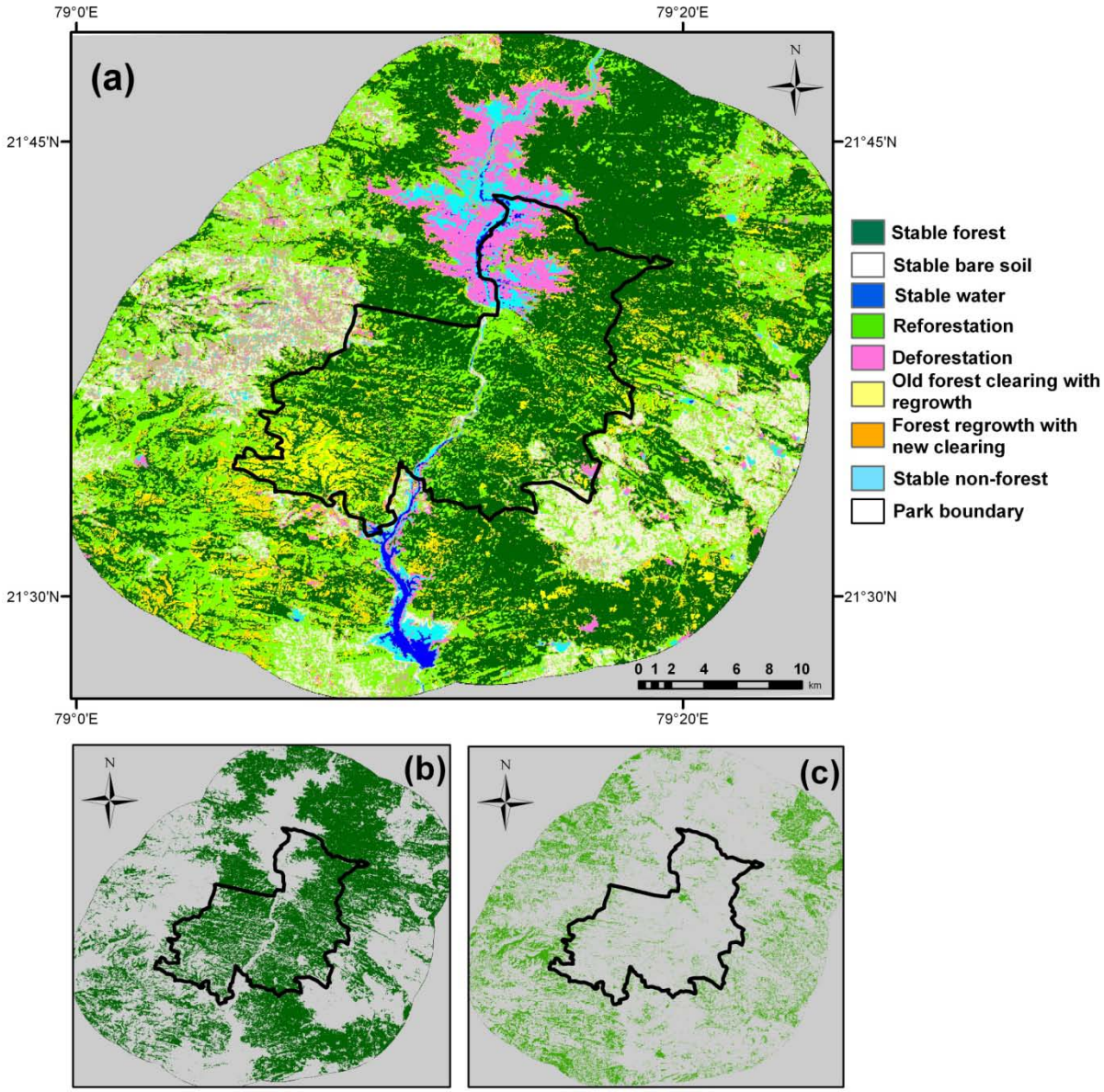


Figure.6-1. Study area showing (a) four-date change trajectory, (b) stable forest mask, and (c) regenerated forest mask.

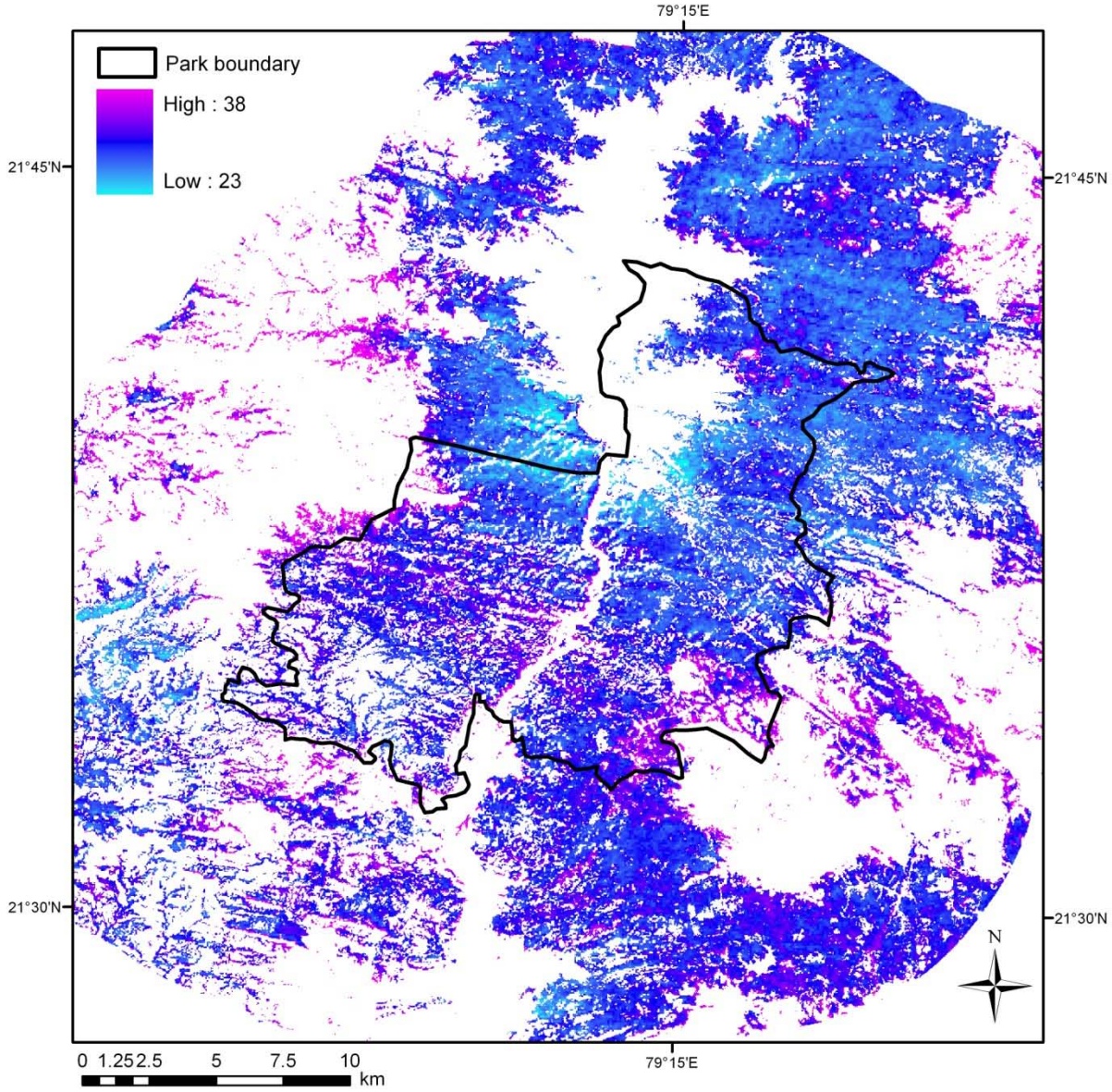


Figure.6-2. Study area showing estimated blackbody surface temperature of stable forest.

CHAPTER 7
CONCLUSION

Contribution of the Study

Tropical forests provide important ecosystem services. In order to protect those forests and forest dependent species, protected areas have been established across the globe. While the field of biodiversity conservation increasingly recognizes the need for empirical evaluations of conservation interventions, few well-designed empirical studies try to show what could have happened in the absence of the conservation efforts. In addition, landscape dynamics in and around a protected area need to be identified, in order to inform natural resource managers. This dissertation targets a tropical protected landscape located in Central India. This decadal study documents changes in land cover in and around PTR during 1977-2007, with emphasis on forest transitions and productivity. It has been suggested by the park managers and the academics that a particular national-level policy change was particularly effective in increasing forest cover in national parks countrywide. This study tests the effectiveness of that particular policy in limiting deforestation in PTR by generating modeled landscapes and then comparing them with the actual landscapes. Since tropical forests tend to saturate in terms of the most widely used remote sensing vegetation index NDVI, this study utilizes EVI2 in quantifying land cover diversity in the study area. This study also reports the capability of ASTER-derived surface temperature, emissivity and tasseled cap indices in detecting forest successions. While reporting regional changes through interdisciplinary tools and techniques, this study also highlights the applicability and potential of some of the newly developed indices and methods in landscape level studies. The overall focus of this research is environmental geography with specific

contribution towards the better understanding of spatial distribution of human-environmental interactions. Broadly this research contributes to two related fields of study – land change science and biodiversity conservation/natural resource management.

Implications for PTR

PTR, as one of the tiger reserves of India, holds potential to sustain viable tiger population, in presence of thriving prey base, which in turn depends on healthy vegetation cover. While PTR is the smallest tiger reserve in India, it is contiguous to another tiger reserve with the same name in the state of Madhya Pradesh. In addition, PTR is connected to Kanha Tiger Reserve in Madhya Pradesh. In order to maintain the connectivity with other suitable tiger habitats it was therefore necessary to gain knowledge on the changing landscape dynamics after its establishment as a national park.

PTR witnessed several changes in degree of protectionism after its declaration as a national park in 1975 and as a tiger reserve in 1999. Relocation of core villages and settlements outside PTR, and presence of over 40 villages within 10 km of park boundary have resulted in a highly human modified matrix in which PTR is embedded. Identifying the spatial patterns of changes in land cover would facilitate better understanding of the underlying processes and mitigating them. A new management plan has been approved by the Government of India, where the surrounding has been declared as the buffer area. This declaration along with alternate ways of subsistence livelihood for the local people could ensure decline in anthropogenic pressure in the core and buffer forest. As this study has showed, presence of villages in the surroundings has resulted in a patchy mosaic of land uses, which could be limiting

animal movements. Further efforts should be made at national and sub-national administrative level to ameliorate park-people relationship and to introduce a truly ‘participatory’ management approach.

This study is the building block of a larger-scale study that I intend to pursue in future. The Pench landscape is fascinating for a land change scientist in many ways, and will play a pivotal role in the local chapter of the global tiger conservation effort. Future directions of research would include (a) identifying subtle “within-class” changes using advanced remote sensing techniques, which will contribute to land change science, (b) identifying degradation in non-protected matrix, which will have crucial implications for biodiversity conservation, and (c) developing and testing robust methods from different sub-disciplines applicable to any ecosystem, which will require an inter-disciplinary approach to study human-environmental interactions.

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BIOGRAPHICAL SKETCH

Pinki Mondal was born in 1981 in Calcutta, India. She graduated from high school in 1999. She went to the University of Calcutta to earn her Bachelor of Science degree in 2002 with geology major (honors), and minors in chemistry and mathematics. In 2004, she earned her Master of Science degree from Jadavpur University, Calcutta in applied geology, with specialization in remote sensing. Her Masters thesis revolved around application of remote sensing in measuring land use and land cover change in the vulnerable estuarine ecosystems of Sundarbans, India. She chose to attend graduate school in the US to get better exposure to current developments in the fields of land change science, and remote sensing. While preparing for GRE, she completed the course works required to earn a post-graduate diploma in computer application from the Department of Electronics and Accreditation of Computer Classes (DOEACC) under the Indian Ministry of Communications & Information Technology. In 2006, she started her graduate studies at the University of Florida to earn a PhD degree in geography with land change science concentration and multi-disciplinary minor.