Landscape change in southwest China's Himalayan mountains:

Implications for old-growth forests, alpine meadows, and avian biodiversity

By

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Abstract

Land use and land cover change (LULCC) is the main cause of biodiversity declines worldwide. Many of the remaining high-diversity ecosystems are located in developing countries, which are undergoing rapid development and population growth. High biodiversity often occurs in the same areas where people dwell, and our understanding of how to balance livelihoods and conservation is limited. Population growth, economic development, conservation policies and climate change interact at multiple spatial and temporal scales to form complex LULCC dynamics, with unexpected consequences for biodiversity.

My overarching goal was to identify effective conservation strategies in developing countries. To this end, I focused on a case study in northwest (NW) Yunnan, a biodiversity hotspot in the remote Chinese Himalayas. NW Yunnan is subject to many forces of change acting at multiple scales, including environmental protection policies, rapid economic development, indigenous land use practices, and climate change. I used satellite imagery to investigate the patterns and drivers of land cover change from 1974 to 2009, and I integrated ecological field data to understand how the observed changes influenced biodiversity.

My results showed that the two highest-diversity ecosystems in NW Yunnan – old-growth forest and alpine meadows – underwent rapid land cover changes that have negative implications for biodiversity. First, I studied change in forest ecosystems, which cover approximately 65% of NW Yunnan. I found that after the landmark logging ban in 1998, overall forest cover increased from 62% in 1990 to 64% in 2009. However, clearing of high-diversity old-growth forest accelerated, from approximately 1100 hectares/year before the logging ban (1990 to 1999), to 1550 hectares/year after the logging ban (1999 to 2009). Paradoxically, old-growth forest clearing accelerated most rapidly where ecotourism was most prominent. Increasing forest cover represented mainly non-pine scrub forests, which were not on a trajectory towards becoming high-diversity forest stands.

Second, I analyzed change in alpine meadows, which have exceptionally high species richness, beta diversity, and endemism. I found that, between 1990 and 2009, at least 39% of alpine meadows converted to woody shrubs. The patterns of change suggest that a catastrophic regime shift is occurring, driven by feedback mechanisms involving climate change, environmental policy that prohibited intentional burning and economic development that increased grazing pressure. Shrub expansion threatens alpine meadow biodiversity and local Tibetan yak herder livelihoods, and more generally, these trends serve as a warning sign for the greater Himalayan region where similar vegetation changes could greatly affect biodiversity, livelihoods, hydrology, and climate.

Finally, I studied the role of Tibetan sacred forests for avian biodiversity. I collected breeding bird and habitat data in six Tibetan sacred forest sites and their surrounding matrix. I found that sacred forests protected old-growth forest ecosystems, supported a significantly different bird community than the surrounding matrix, and had higher bird species richness at multiple scales. While bird community composition was primarily driven by the vertical structure of the vegetation, plots with the largest trees (height or diameter) and bamboo groves had the highest bird diversity and abundance, indicating the importance of protecting old-growth forest ecosystems for Himalayan forest birds.

In general, my dissertation shows that complex interactions between environmental policy, economic development strategy, and climate change in tightly coupled human-nature systems can lead to unexpected trajectories of land cover change. Satellite imagery, when paired with ecological field data, can measure these broad-scale changes and their implications for biodiversity, thereby informing policy and management in a timely manner.

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Introduction

Biodiversity loss is a global crisis because it profoundly alters ecological processes and disrupts ecosystem services. As humans take up an ever-increasing proportion of the Earth's resources, biodiversity continues to decline (Chapin et al. 2000; Myers & Kent 2003; Dobson et al. 2006). Land use and land cover change (LULCC) is one of the most important factors threatening species diversity, because it changes species habitat. However, many aspects of the relationship between biodiversity and LULCC are not well understood.

For example, the species-area relationship derived from island biogeography (MacArthur & Wilson 1967) asserts that the larger an area, the higher species diversity it will support. But this is not universally true. At broad spatial scales, there are "biodiversity hotspots", or regions that contain disproportionately high levels of biodiversity (Myers et al. 2000). At finer scales, there are "keystone structures", discrete spatial features that maintain biodiversity despite being small in proportion to the entire ecosystem (Tews et al. 2004). In an increasingly human-dominated world, protecting biodiversity hotspots and identifying keystone structures is critical for conservation.

Many biodiversity hotspots are located in developing countries (Myers et al. 2000; Zimmerer et al. 2004). Developing countries are undergoing rapid development and population growth, government institutions are often weak, and enforcement levels are low. Furthermore, biodiversity is frequently located in the same areas where people dwell, creating conflicts between livelihoods and conservation (Naughton-Treves et al. 2005; Wittemyer et al. 2008; Ferraro et al. 2011). For effective environmental management, it is thus essential to align conservation and development goals, but our understanding of how to balance conservation and livelihoods is still limited (Ferraro et al., 2011).

The challenge is that development typically involves land use change but the dynamics of land cover change are increasingly complex, and difficult to predict. More and more of Earth’s resources have been appropriated for human use, with negative consequences for biodiversity (Foley et al. 2005). Although intensification of human land use does often result in direct losses of habitat, simplistic explanations of the drivers of LULCC, such as population growth and poverty, are inadequate (Lambin et al. 2001). LULCC is a result of people’s responses to economic opportunities, which in turn result from complex interactions of political, social and economic drivers acting at multiple scales (Lambin & Meyfroidt 2010).

As economies have changed, novel LULCC trajectories have become increasingly common (Hostert et al. 2011). For example, more and more countries have undergone a forest transition, i.e., a change in forest trajectory from decreasing to increasing forest cover (Meyfroidt et al. 2010; Meyfroidt & Lambin 2011). Forest transition theory assumes that as a country develops, its forest trajectory follows an environmental Kuznets curve, i.e., the environment first worsens but then improves as incomes rise (Mather et al. 1999). The opportunities of returning forests for carbon sequestration, climate and hydrologic cycles, and biodiversity conservation are substantial (Rudel et al. 2005; Kauppi et al. 2006), yet we have very little understanding of the patterns, drivers, and consequences of forest transition in developing countries (Chazdon 2008; Chazdon et al. 2009; Meyfroidt & Lambin 2009; Perfecto & Vandermeer 2010).

Another LULCC dynamic that is not well-understood are regime shifts (Scheffer 2009). Regime shifts occur when multiple interacting factors reach a critical threshold, triggering unexpected, and potentially irreversible, transitions to an alternate state. Predicting critical thresholds and identifying contributing factors is necessary to avoid ecosystem shifts, but untangling these factors is challenging in coupled natural and human systems (Liu et al. 2007; Brock & Carpenter 2010).

One key to understanding forest transitions, regime shifts, and conservation in biodiversity hotspots is to monitor land cover change over broad spatial and temporal scales in response to climate changes, protection strategies and economic development. Remote sensing is a crucial tool for land cover change mapping and monitoring. It provides a tool to study landscape pattern and understand its influence on ecosystem processes and biodiversity at spatial and temporal scales that are otherwise impossible to study (Nagendra 2001; Turner et al. 2003).

Remote sensing is an especially important monitoring and conservation tool in developing countries. Official records of land use and land cover are limited, and thus satellite imagery is often the only source of historical data. In addition, remote and rugged landscapes inhibit field data collection, making satellite imagery the best source of consistent data across broad spatial scales. Despite its importance, there are real challenges associated with remote sensing analyses in developing countries, including the limited availability of satellite, ground-truth and biological field data to train and validate remote sensing algorithms. Therefore, applied research that tests the potential, and the limits, of remote sensing to achieve accurate and meaningful land cover change monitoring and biodiversity assessment in developing countries is crucial.

The overarching goal of my research was to identify effective strategies for biodiversity protection in developing countries. To reach this goal, I addressed broad questions of LULCC, biodiversity, and conservation in a highly diverse, rapidly changing, and little-studied region of Southwest China. My study area was NW Yunnan (Figure 1), a global biodiversity hotspot (Myers et al. 2000) and a UNESCO World Heritage site. NW Yunnan is still relatively undeveloped but experiencing rapid change. Local peoples continue to practice subsistence-based agriculture and pastoralism, but since the 1970s, NW Yunnan has undergone major changes due to national policies aimed at fostering both economic development and environmental protection. These policies stimulated rapid infrastructure development, immigration of culturally-dominant Han Chinese, tourism, new protected areas, and changes in land use. In addition, NW Yunnan is experiencing accelerated climate change (Baker & Moseley 2007a). However, the implications of these diverse factors for land cover change and biodiversity are largely unknown.

My research characterized LULCC patterns, and the consequences of LULCC for NW Yunnan's unique floral and faunal diversity. I used remote sensing coupled with field-based measurements, and statistical modeling to address my research questions. My dissertation consists of an introduction, which provides a summary of the entire dissertation, and three chapters, which examine specific research questions in detail.

Chapter 1 Summary

Research Question: How did forest ecosystems change from 1974 to 2009 in response to economic and environmental protection policies?

Hypothesis: Logging decreased following the 1998 logging ban.

NW Yunnan's temperate forest ecosystems are the most biologically diverse temperate forests globally and the primary target of conservation efforts in this biodiversity hotspot. In 1998 the Chinese government instituted the National Forest Protection Plan (NFPP), whose primary objective was to ban logging of all forests in southwest China, except for small quotas allowed to local people for non-commercial uses. Furthermore, since the 1990s China has invested heavily in reforestation programs and ecotourism.

In order to understand how forest ecosystems changed in response to these various protection and development policies, I used Landsat TM/ETM+ images and multi-temporal change classification to analyze forest change in three time periods: Historic (1974 – 1990), Pre-ban (1990 – 1999) and Post-ban (1999 – 2009). I summarized forest loss and regeneration for the three time periods, and at three different scales (entire study area, county-level, and township-level).

My results showed that total forest cover increased from 1974 to 2009. However, the logging of high-diversity old-growth forest accelerated, especially in forests around Shangrila. Old-growth forest logging accelerated because of high demand for old-growth timber due to rapid ecotourism-based economic development of Shangrila. Much of the new construction for tourists (guesthouses, tourist attractions, restaurants, and vacation homes) uses old-growth forest timber, and increasing wealth of local people resulted in more – and larger – house construction.

Increasing forest cover represented mainly non-pine scrub forests, which were not on a trajectory towards becoming high-diversity forest stands. The accelerated loss of high-diversity old-growth forest and the proliferation of scrub forests indicated that despite increasing forest cover, forest biodiversity in NW Yunnan likely continued to decline. My results highlighted that forest monitoring must incorporate multiple forest classes to assess forest change in the context of the conservation of biodiversity and ecosystem services. Simple forest versus non-forest cover assessments in areas with remaining unprotected old-growth forests are inadequate to understand the implications of protection and development strategies for high-diversity forest types, and can obscure important environmental degradation processes.

Ecotourism, or nature-based tourism, is expanding around the world because of increasing demand, and because it may offer a strategy to wed sustainable economic development with environmental protection (Balmford et al. 2009; Karanth & DeFries 2011). However, my research suggests that rapid development may pose inherent risks to biodiversity given that our study area, with strong forest protection policies and ecotourism-based development, arguably represents a "best-case scenario" for balancing development with maintenance of biodiversity.

Resulting paper: Brandt, J. S., T. Kuemmerle, H. Li, G. Ren, J. Zhu, and V. C. Radeloff. 2012. Using Landsat imagery to map forest change in southwest China in response to the national logging ban and ecotourism development. Remote Sensing of Environment 121: 358-369

Chapter 2 Summary

Research question: What are the rates and drivers of shrub expansion into alpine meadows in Northwest Yunnan?

Hypothesis: Shrub encroachment was driven by cessation of controlled, intentional burning in alpine shrublands since the 1998 burning ban.

Historically, Himalayan alpine meadows have been used by indigenous agro-pastoralists whose rangeland management practices sustained both local livelihoods and biodiversity (Klein et al. 2011). Since the 1950s, numerous environmental changes have affected Himalayan ecosystems. Climate warming rates are higher than the global average, changing ecosystem productivity (Peng et al. 2010), phenology (Yu et al. 2010b), and permafrost (Xu et al. 2009). Changing political and social structures have led to economic development, and abandonment of traditional land use practices (Yeh & Gaerrang 2010; Klein et al. 2011). However, despite grassland degradation throughout the Himalayas (Harris 2010), rates of shrub encroachment have not been quantified, and its drivers are not well understood.

The alpine meadows of NW Yunnan have higher species richness, beta diversity, and endemism than elsewhere in the Himalayas (Salick et al. 2009; Xu et al. 2009). Their disappearance represents a threat to both biodiversity and local livelihoods. I used Landsat TM/ETM+ satellite image analysis and dendrochronology to map shrub encroachment in the alpine zone (3,800 - 4,500 m) from 1974 to 2009. In addition, I reconstructed changes since the 1950s using interviews, livestock records, and climate data.

Results showed that alpine meadows remained largely unchanged from 1974 to 1990 despite changes in climate and land use, but an abrupt shift occurred from 1990 to 2009 when at least 39% of all meadows were encroached by shrubs. This represents a very rapid encroachment rate (2.1% per year) higher than rates reported elsewhere (Briggs et al. 2005; Coop & Givnish 2007; Thompson 2007; Sankey & Germino 2008). Given the exceptionally rich flora with many endemics, this shrub encroachment is likely to seriously deplete plant diversity in the region (MacArthur & Wilson 1967) and represents a major conservation threat.

The patterns of shrub encroachment suggested that the alpine meadows of NW Yunnan are undergoing a regime shift from an herbaceous to a shrub-dominated ecosystem. Despite multiple perturbations to the climate and land use systems starting in the 1950s, alpine meadows remained resilient to shrub expansion until the late 1980s. Thereafter, rhododendron establishment across several alpine territories appears to correspond to two potential triggers. First, May temperatures at 4000 m shifted from sub-zero to above-freezing levels in the mid 1980s, likely reducing snowpack and snowfall. Second, the 1988 burning ban immediately reduced burning by about 20%, providing another potential trigger. Once established, shrublands rapidly expanded due to feedback mechanisms involving climate, woody cover, and grazing. Shrubs spread most rapidly in areas with some prior woody vegetation (shrub autocatalysis) and areas without spring snow cover. Burning historically controlled shrubs, and as intentional burning decreased, shrubs expanded. As shrubs expanded, remaining meadows were subject to overgrazing, promoting further shrub expansion.

The results call for immediate, adaptive management in alpine ecosystems of NW Yunnan. The regime shift poses a serious threat to both endemic meadow biodiversity and local livelihoods. Traditional herding systems sustained both local livelihoods and endemic biodiversity. However, the socio-economic and climatic conditions in NW Yunnan have changed greatly. Grazing and burning were compatible with sustaining alpine ecosystems in the past, but may be less sustainable now given climate change and heavy shrub encroachment. Furthermore, our study area may represent a sentinel for the rest of the greater Himalayan region, where similar vegetation changes could greatly affect livelihoods, hydrology, and climate.

Related paper (In review): Brandt, J.S., M.A. Haynes, T. Kuemmerle, Fang Zhendong, D. Waller, and V. C. Radeloff. Regime shift on the roof of the world: Alpine meadows convert to shrublands in the southern Himalayas. *Biological Conservation.*

Chapter 3 Summary

Research question: What is the ecological and conservation role of sacred forests for forest birds?

Hypothesis: Sacred forests are keystone structures for avian diversity in NW Yunnan.

In Chapter 1, I found that logging of old-growth forests accelerated in recent times, and non-pine scrub forests have become more widespread. This raises the question how these changes affected forest bird communities.

Although our study area is within a center of avian endemism in China (Lei et al. 2003), no research had been done on forest songbird distribution, diversity, population trends, or habitat selection. Sacred forests retain old-growth forests, and these forests and their surrounding matrix together encompass a wide range of forested habitats, providing a convenient study design to understand 1) what birds are breeding in different forest types of NW Yunnan, 2) what habitat characteristics the birds are selecting, and 3) the ecological and conservation role of sacred forests for avian diversity in our study area.

To answer these questions, I surveyed birds and their habitat in and around six Tibetan sacred forest patches over two years. I found that sacred forests supported a significantly different bird community than the surrounding matrix, had higher bird abundance, and higher bird species richness at the plot, patch and landscape scales. Furthermore, while I encountered a single matrix bird community, the sacred forest patches provided high within and between-patch heterogeneity, and supported multiple distinct sacred-forest bird communities. I also identified habitat characteristics important for forest birds. While bird community composition was primarily driven by the vertical structure of the vegetation, plots with the largest trees (height or diameter) and bamboo groves had the highest bird diversity and abundance, indicating the importance of protecting old-growth forest ecosystems for Himalayan forest birds.

Old-growth forest clearing and degradation have accelerated throughout the Himalayas, and our results offer hope for forest birds, because they indicate that existing sacred areas protect a variety of habitat niches and increase avian diversity at multiple spatial scales. As population growth and rapid economic development continues throughout the Himalayas, sacred sites are an important opportunity for biological conservation at the local, landscape and regional scales.

Related paper (In prep.) Brandt, J. S., Han Lianxian, Fang Zhendong, E. M. Wood, A. M. Pidgeon, V. C. Radeloff. Sacred forests are keystone structures for forest bird conservation in southwest China's Himalayan mountains. *Conservation Biology*.

Significance of my dissertation

My research addressed broad themes of LULCC, biodiversity, and conservation, with NW Yunnan as my case study. NW Yunnan represents a unique opportunity to study linkages between LULCC and biodiversity in developing countries, yet very little is known about the region because access for foreign scientists was limited. What makes the region particularly interesting though is that China has implemented over the last 30 years several strong social, economic and environmental policies in NW Yunnan that led to dramatic changes in human land use practices. During the same time period, the region has experienced climatic change greater than the global average. Measuring land cover changes at broad temporal and spatial scales allowed me to understand the implications of and interactions among policies, climate change and development strategies. What I learned in NW Yunnan provides an opportunity to gain insights to future LULCC dynamics in other developing regions world-wide, and my dissertation makes scientific contributions in three dimensions: to ecology, to remote sensing and to conservation.

My dissertation advances **ecological** knowledge. My research focused on two unique and relatively unknown ecosystems, temperate forests and alpine ecosystems in the Himalaya. Gathering baseline information on relatively unknown ecosystems is important in and of itself, and a necessary first step on which to base future ecological research. My research on forest birds described avian communities and their habitat selection in NW Yunnan forests, which had not yet been done. Furthermore, the bird research highlighted the ecological differences between remnant old-growth forest patches, forest edges, and disturbed forest ecosystems for bird communities in NW Yunnan. In alpine meadows, I identified an unexpected and previously undocumented regime shift from herbaceous- to shrub-dominated ecosystem, attributable to interactions between changes in land use and climate. Little is known about woody encroachment of grasslands in Asia, and my research highlights that rapid shifts can occur in sensitive high-elevation ecosystems previously adapted to traditional land management.

I made advances in **technical approaches** that are relevant to a wider remote sensing audience. I used multi-temporal image analysis to overcome the challenges inherent for remote sensing analysis of mountainous ecosystems with a monsoonal climate. I used Support Vector Machines (SVM), a machine-learning algorithm only recently adapted for multi-class image analysis, and I tested the ability of SVM to tackle multi-class problems with non-random training data. Furthermore, in such a remote and data-poor study area, remotely-sensed data alone was not sufficient to map LULCC and understand factors contributing to that change. Pairing remote sensing with field and interview data, I accurately characterized relatively subtle, but biologically significant, changes among land cover types. I integrated economic, social and ecological data into the remote sensing analysis to analyze LULCC patterns and explore hypothesized drivers of the observed changes.

Finally, my research is highly relevant for **conservation**. I investigated the consequences of forest transitions, ecotourism expansion, and climate change, all of which are global biodiversity conservation issues. In the case of forest ecosystem change, I identified unexpected acceleration of old-growth forest logging due to ecotourism-based development in NW Yunnan. Ecotourism is expanding around the world, but my research suggests ecotourism-based development is not in all cases a panacea for economic development and environmental protection. In the alpine ecosystems of NW Yunnan, I identified rapid rates of shrub expansion, which threatens alpine meadow biodiversity and local Tibetan yak herder livelihoods. More generally, my study area could be a sentinel for alpine ecosystems of the Greater Himalayas and world-wide, where similar vegetation changes could greatly affect biodiversity, livelihoods, hydrology, and climate. Finally, my research highlighted that sacred forests are promising keystone structures for avian diversity in this region, and perhaps in the greater Himalayas. In an increasingly human-dominated world, the identification of keystone structures and understanding how they function is critical for conservation.

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Chapter 1: Using Landsat imagery to map forest change in southwest China in response to the national logging ban and ecotourism development

Abstract

Forest cover change is one of the most important land cover change processes globally, and old-growth forests continue to disappear despite many efforts to protect them. At the same time, many countries are on a trajectory of increasing forest cover, and secondary, plantation, and scrub forests are a growing proportion of global forest cover. Remote sensing is a crucial tool for understanding how forests change in response to forest protection strategies and economic development, but most forest monitoring with satellite imagery does not distinguish old-growth forest from other forest types. Our goal was to measure changes in forest types, and especially old-growth forests, in the biodiversity hotspot of northwest Yunnan in southwest China. Northwest Yunnan is one of the poorest regions in China, and since the 1990s, the Chinese government has legislated strong forest protection and fostered the growth of ecotourism-based economic development of the region. We used Landsat TM/ETM+ and MSS images, Support Vector Machines, and a multi-temporal composite classification technique to analyze change in forest types and the loss of old-growth forest in three distinct periods of forestry policy and ecotourism development from 1974 to 2009. Our analysis showed that logging rates decreased substantially from 1974 to 2009, and the proportion of forest cover increased from 62% in 1990 to 64% in 2009. However, clearing of high-diversity old-growth forest accelerated, from approximately 1100 hectares/year before the logging ban (1990 to 1999), to 1550 hectares/year after the logging ban (1999 to 2009). Paradoxically, old-growth forest clearing accelerated most rapidly where ecotourism was most prominent. Despite increasing overall forest cover, the proportion of old-growth forests declined from 26% in 1990, to 20% in

2009. The majority of forests cleared from 1974 to 1990 returned to either a non-forested land cover type (14%) or non-pine scrub forest (66%) in 2009, and our results suggest that most non-pine scrub forest was not on a successional trajectory towards high-diversity forest stands. That means that despite increasing forest cover, biodiversity likely continues to decline, a trend obscured by simple forest-versus-non-forest accounting. It also means that rapid development may pose inherent risks to biodiversity, since our study area arguably represents a “best-case scenario” for balancing development with maintenance of biodiversity, given strong forest protection policies and an emphasis on ecotourism development.

Introduction

Land cover and land use change are the main causes of biodiversity declines (Vitousek et al. 1997; Chapin et al. 2000; Foley et al. 2005), and old-growth forests are among the most threatened habitats globally. Old-growth forests are economically valuable for timber (Chazdon et al. 2009) and as agricultural land (Gibbs et al. 2010; Perfecto & Vandermeer 2010), and they continue to disappear despite many efforts to protect them. Most of the remaining high-diversity old growth forests are located in developing countries (Myers et al. 2000; Zimmerer et al. 2004), which are undergoing rapid development and population growth. High biodiversity often occurs in the same areas where people dwell (Naughton-Treves et al. 2005), and our understanding of how to balance livelihoods and conservation is still limited (Ferraro et al. 2011).

However, while old-growth deforestation continues, more and more countries have undergone a forest transition, i.e., a change in forest trajectory from decreasing to increasing forest cover (Meyfroidt et al. 2010; Meyfroidt & Lambin 2011). Forest transition theory assumes that as a country develops, its forest trajectory follows an environmental Kuznets curve, i.e., the environment first worsens but then improves as incomes rise (Mather et al. 1999). While the opportunities of returning forests are substantial (Rudel et al. 2005; Kauppi et al. 2006; Rudel 2009), increasing forest cover alone does not necessarily mean that biodiversity and natural ecosystems are on a pathway towards recovery. Secondary forests are not equal to old-growth forest in terms of biodiversity, carbon storage, and ecosystem service provision (Chazdon 2008; Rudel 2009; Perfecto & Vandermeer 2010) and do not always return to high-diversity ecosystems (Chazdon et al. 2009).

As old-growth forests dwindle and new forests proliferate, two key challenges are to 1) identify effective protection strategies for remaining old-growth forests, and 2) understand the fate and implications of the increasing area of new forests. The key to both is to monitor forest change dynamics over broad spatial and temporal scales in response to different protection strategies, government policies, and economic development.

Remote sensing is a crucial tool for forest cover change mapping and monitoring. Mapping old-growth forest distribution (Congalton et al. 1993) and post-disturbance forest succession (Fiorella & Ripple 1993; Cohen et al. 1995; Jakubauskas 1996) have long been recognized as essential components of forest biodiversity assessment, and in recent years, multiple forest classes have been mapped even in extremely complex and little-studied environments (Helmer et al. 2000; Liu et al. 2002; Schmook et al. 2011).

However, detailed forest type classifications are usually performed for only a single time period, because there are formidable challenges associated with mapping change for multiple classes over several timesteps. First, using single-date classifications to detect change (i.e. post-classification change detection analysis) over multiple timesteps is problematic because errors multiply over each timestep (Kennedy et al. 2009). Composite change detection minimizes multiplicative error by stacking multi-date imagery together and classifying change directly. However, change classes typically have non-normal distributions. For example, even though deforested areas transition to grassland, agriculture, bare land, or shrub, all are included into a single “deforestation” class, and most classification techniques are ill suited to handle such complex class distributions.

Recently, non-parametric classification techniques, such as decision trees (Hansen et al. 2008; Potapov et al. 2011) have been tested for composite change detection because they can accommodate the non-normal and multi-modal distributions of multi-date imagery. The major disadvantage of non-parametric techniques for composite change detection is that they typically perform better with training datasets that provide a complete and representative sample of the classes (Pal & Mather 2003). In composite change detection, overall change class numbers increase exponentially with each added land cover class, and adequate training data for rare land cover classes over multiple time-steps are difficult to acquire (Kennedy et al. 2009).

Support Vector Machines (SVM) are an alternative non-parametric classifier that offer particular promise for change detection of multiple forest classes because they can handle complex distributions of multi-temporal imagery (Huang et al. 2002), but they do not require training datasets that completely describe each class. SVM place hyperplanes to separate different classes, and only training points at the class boundaries are necessary for optimal hyperplane placement (Foody & Mathur 2004). Therefore, SVM can perform effectively with a small sample of “mixed pixels” collected from purposefully selected locations (Foody et al. 2006).

Our overarching goal was to use remote sensing to map different forest types and forest loss in complex environments in order to understand processes affecting high-diversity forest types. Our study area was Diqing Prefecture of northwest (NW) Yunnan Province in the Himalayan mountains of southwest China, a global biodiversity hotspot (Myers et al. 2000) and rapidly developing region. Home to the most biologically-diverse temperate forests in the world (Morell 2008), and historically relatively undisturbed (Goodman 2006), large expanses of NW Yunnan’s old-growth forests were clear-cut by state logging companies from the 1960s through the 1990s to fuel China’s national development (Harkness 1998; Morell 2008) and the logging industry dominated the local economy (Melick et al. 2007).

However, in response to catastrophic flooding along the Yangtze River, in 1998 the Chinese government instituted the National Forest Protection Plan (NFPP). One of the primary objectives of the NFPP was to ban logging of all forests in southwest China, except for small quotas allowed to local people for non-commercial uses (e.g. construction materials and fuelwood). Furthermore, since the 1990s China has invested heavily in reforestation programs (Liu et al. 2008) and ecotourism (Kolas 2008; Jenkins 2009). As a result, forest cover in SW China is increasing (Weyerhauser et al. 2005), but fine-scale studies indicate that old-growth forests continue to be logged (Melick et al. 2007; Xu & Melick 2007; Zackey 2007) and the ecological integrity of the new forests is unclear (Liu et al. 2008; Xu 2011).

Using remote sensing to map forest change is a logical first step towards understanding the consequences of forest protection and economic development policies since the 1980s in SW China. However, remote sensing in the region is challenging. Because of the monsoonal climate, clouds cover the region during the growing season, but winter image analysis is challenged by snow cover, illumination effects from topography, and senescent vegetation. Furthermore, the collection of ground-truth data is difficult because topography is extremely rugged and roads are few, aerial photos are not freely available, and different forest types are difficult to separate visually in either Landsat or high-resolution imagery.

To overcome these challenges, and to understand complex processes of forest change in our study area, we used SVM, multi-temporal Landsat TM/ETM+ satellite imagery, purposefully-selected ground truth data, and a combined post-classification and composite change detection technique to map multiple classes of forest cover and change in NW Yunnan from 1974 to 2009. Our specific objectives were to:

1. Map multiple forest classes and forest loss for the historical period (1974-1990), the decade before the logging ban (1990-1999) and the decade after the logging ban (1999-2009).

2. Assess the spatial and temporal patterns of logging in relation to geographic, demographic, and economic factors.
3. Determine overall forest cover change, and types of forest that have regenerated.

Study Region

Our study area (22,834 km²) was the Diqing Tibetan Autonomous Prefecture in the Hengduan Mountains of northwest (NW) Yunnan Province, bordering Tibet and Sichuan Province (Fig. 1a). Elevations in the study area range from 1500 to 6000 m above sea level (Fig. 1b), creating a large array of ecological niches in a relatively small area. Forest cover in our study area has been estimated at 60% (Weyerhauser et al. 2005), but the high-diversity old-growth forests, which are the primary conservation target in this biodiversity hotspot, are just a fraction of the total forest cover. The old-growth montane conifer and mixed forests are the most biologically diverse temperate forests globally (Morell 2008). Over 7,000 plant, 410 bird, and 170 mammal species have been documented in NW Yunnan, many of which are endemic to native old-growth forests (Xu & Wilkes 2004; Chang-Le et al. 2007; Ma et al. 2007). Among the different land cover types, the old-growth forest community is richer in endemic, endangered and culturally useful species than any other land cover types (Wen et al. 2003; Anderson et al. 2005; Ma et al. 2007; Salick et al. 2007; Li et al. 2008; Wang et al. 2008), and is essential habitat to the endangered Yunnan snub-nosed monkey (Wen et al. 2003; Li et al. 2008), and to several rare species of pheasants (Wang et al. 2008).

NW Yunnan is also an UNESCO world heritage site because of its centuries-long history of indigenous subsistence cultures, including Tibetan, Lisu, Bai, Naxi, and Yi peoples. Historically, the region was sparsely populated (approximately 15 people/km^2), with a decreasing population gradient from the lower-elevation South to the higher, harsher-climate

North (Fig. 1c). Most people live at a subsistence level, relying heavily on forests for their livelihoods. Forests are still the primary source of fuel for cooking, heating, and construction, and are intensively used for livestock grazing, hunting, food gathering, and traditional medicines. Old-growth trees are especially valuable to Tibetans, as large logs are required to construct traditional Tibetan houses.

NW Yunnan is one of China's poorest regions, making it a primary target of economic development programs since the 1980s, including the Great Western Development program in 1998, which emphasizes infrastructure development, ecological rehabilitation, foreign economic investment, and education throughout western China (Xu et al. 2006). Furthermore, tourism development in NW Yunnan's natural areas has been promoted as a strategy for both environmental protection and economic development (Li & Han 2000; Wang & Buckley 2010).

Methods

General Approach

We used Landsat MSS/TM/ETM+ images and a multi-temporal change classification technique to map forest cover and change in NW Yunnan from 1974 to 2009. We focused on the classification of forest cover loss and of three biologically distinct forest types. A field-derived training dataset and multitemporal Landsat imagery (Fig. 2) enabled accurate classification of the three different forest classes. We performed a combination of composite and post-classification change detection, using multi-temporal imagery from four different time periods (1974, 1990, 1999 and 2009) to quantify land cover/land use change during three intervals (1974-1990, 1990-1999, and 1999-2009) (Fig. 3). One composite change classification mapped forest types and forest loss in the 'Historic' time period (1974-1990), and another mapped forest types and forest loss in both the 'Pre-ban', i.e., the decade before the logging ban (1990-1999), and 'Post-ban', i.e., the decade following the logging ban (1999-2009) periods. We then performed post-classification change detection from the two composite change classification to identify which types of forests were logged in the Pre-ban and Post-ban periods, and to measure what was the share of total forest cover in 2009 that had been logged in the Historic period. The three forest types were:

(a) Old-growth forests represent the main target of biodiversity conservation in this region (Xu & Wilkes 2004; Chang-Le et al. 2007; Ma et al. 2007). We define old-growth forests as the native, original forest vegetation community of this region. The old-growth forest community is composed of mixed evergreen and deciduous species, including fir (*Abies* spp.), spruce (*Picea* spp.), pine (*Pinus* spp.), larch (*Larix* spp.), evergreen oak (*Quercus* spp.), birch (*Betula* spp.) and rhododendron (*Rhododendron* spp.) with specific species composition highly related to topographic variability (Li & Walker 1986b). The vast majority of this category consists of old-growth forest vegetation community in its climax state, but also includes this community in its secondary state, because the primary and secondary states are spectrally indistinguishable with our Landsat imagery because of the similarity in species composition.

(b) Pine/oak woodlands can occur naturally in NW Yunnan, and often colonize and persist in cleared areas after a disturbance (Li & Walker 1986b), but many pine forests have been planted after logging. In our study area, old-growth pine/oak woodlands are rare, and existing pine forests are typically homogeneous stands of young pines (*Pinus densata*) with oak shrub understory.

(c) Non-pine scrub represents a mix of deciduous and evergreen shrublands that regenerate and persist following logging. This forest type is especially common near villages and along roads, where forests are heavily used by livestock and people. Based on interviews with local people, many scrub forests have persisted for many decades after logging.

Satellite images

The study area is covered by two Landsat TM image footprints (path/row 132/040 and 132/041). We used a total of 14 images (Table 1), all from late October through early April, because cloud free images are not available during the growing season in this region due to the monsoonal climate. Images were obtained from the USGS Landsat archives and from the China and Thai International Ground Stations. We georeferenced all images to the Landsat 2000 GeoCover Dataset using ERDAS IMAGINE AutoSync, which was already orthorectified, and used a gap-filled Digital Elevation Model (DEM) from the Shuttle Radar Topography Mission (SRTM) to account for relief displacement. Root mean square error for the georeferencing was less than 0.4 pixels (<12 m). Clouds and cloud shadows were masked manually from the images prior to all image analyses. We did not apply any other pre-processing of the images.

Field data

Field data for training and validation were collected during six months of field work, from September to November of 2008, and August to October of 2009. We sampled at least 40 locations for each land cover class to aid the interpretation of spectral subclasses during image analysis. The ground truth dataset consisted of 1,573 polygons containing 93,300 pixels (Table 2). Due to the ruggedness of the study area, a random sampling design was not feasible. Ground truth data for uncommon classes, such as old-growth forests, required extensive trekking in remote locations. We hired local villagers to guide us to old-growth forests, alpine pastures, and areas of logging, and while trekking we conducted interviews with our guides about land use history and practices. We recorded land use and land cover observations and took photographs for an approximate 100 x 100 m area surrounding each GPS point (Justice & Townshend 1981). We collected points for both “pure pixels” (i.e., homogeneous areas of a single land cover type), as well as “mixed pixels”, (i.e. areas were a mixture between land cover types prevails) to aid the SVM in hyperplane placement.

We supplemented field data with ground-truth data from high resolution imagery available in Google Earth$^{\text{TM}}$, which covered approximately one-third of the study area, and the Landsat imagery. For the forest type classes, we relied entirely on field data, since these classes cannot be distinguished reliably based on visual inspection of single-date satellite imagery alone. For forest loss classes, approximately 60% of training data were derived from field data, and approximately 40% were selected by visual interpretation of the Landsat imagery. We identified forest loss classes in the field by visiting areas of past logging, and asking our local guide when the logging had occurred. Similarly, approximately 20% of the training areas for agriculture, grassland, sparse shrub, and bare/urban classes were collected from the high-resolution imagery, and 80% in the field.

Change detection

We used Support Vector Machines (SVM), implemented in the software imageSVM (Janz et al. 2007), for our change detection. To train and validate our SVM, we used a random selection of approximately 1000 points from each class in our ground truth dataset. We performed a classification for the Historic period (1974-1990) by combining 1974 Landsat MSS images, ca. 1990 Landsat TM images, and elevation and hillshade from the DEM. The hillshade for the ‘Historic’ classification was calculated with a sun elevation angle of 34.04°, and a sun azimuth angle of 146.62° (corresponding to the November 19th 1990 Landsat TM image (Path/Row 132/40). For the 'Historic' classification, all images were resampled to the resolution of the MSS image (57 m pixels). We classified six classes, including the three forest types, a forest cover loss 1974-1990 class, agriculture and grassland, and other (bare/urban/snow/water).

Using the Landsat TM/ETM+ imagery from 1990, 1999, and 2009, we performed a second composite classification for the decade leading up to the logging ban (Pre-ban, 1990-1999), and the decade following the logging ban (Post-ban, 1999-2009). We stacked the five TM/ETM+ images along with elevation and hillshade data. The hillshade image for the Pre-ban and Post-ban classifications was derived using the image acquisition time of the October 28, 1999 image, Path/Row 132/40 (sun elevation = 41.87°, azimuth = 147.17°). One hillshade image was sufficient to represent shading caused by topography. For this classification, all images were resampled to a spatial resolution of 28.5 m. We classified the image stack into 11 classes, including three permanent forest types and forest loss classes in different periods (1990-1999 and 1999-2009), agriculture, grassland, alpine shrub expansion, sparse shrub, bare/urban, and other (snow/water) . To eliminate isolated misclassified pixels, we identified contiguous groups of pixels (using the 4-neighbor rule) and merged small patches (<2 pixels for the Historic classification and <6 pixels for the Pre-ban and Post-ban classification) into the largest neighboring patch (minimum mapping unit of 0.65 ha for the historic classification and 0.49 ha for the Pre-ban and Post-ban classification).

We randomly withheld 10% of the dataset for the accuracy assessment and 90% of the pixels were used for training. We classified the image stack a total of ten times, with a different random training dataset each time, and the final accuracy measures are derived from the mean error estimates of all ten classifications. The final classification was produced from 100% of the data points.

For accuracy assessment, from the confusion matrix we calculated an area-adjusted error matrix, including area-adjusted user, producer, and overall accuracies, that take into account the areal proportions of each class (Card 1982). This is necessary to correct for potential bias due to the differences in the proportions of classes in the validation data and the true areal proportions of these classes in the map. We also adjusted the total area estimates from the classified map according to the bias correction, to produce an adjusted area coverage for each class, and then calculated 95% confidence intervals for these area estimates (Cochran 1977; Card 1982) to provide a more accurate and intuitive representation of error.

Analyzing logging rates and patterns

We compared forest change among the three different time periods, and at three different scales: the entire study area, county-level, and township-level. We used two different measures of forest disturbance. First, we calculated the number of hectares of forest loss. Second, we calculated an area and time-adjusted disturbance rate (Eq. 1)

$$ADR = ((D_j/FCB_j)/t)*100 \quad Eq.\ 1$$

where ADR is the annual disturbance rate, Dj is the number of pixels in the forest cover loss class for time period j, $FCBj$ is the total number of forest pixels at the beginning of time period j, and t is the number of years in time period j.

To understand the influence of humans on forest change patterns, we used spatial data on road networks, villages, and provincial and township boundaries, all digitized from 1:250,000 topographic maps from the late 1990s. We calculated township-scale village density (as a proxy for population density, which was not available), and road density (including national, provincial, county and village-level roads). Pearson correlation and multiple linear regression analyses were performed to determine the relationship between logging rates, village density, and road density during the three time periods. To understand the economic and demographic implications of NW Yunnan's rapid development, we gathered economic statistics from official Chinese sources, compiled at the scale of Diqing Prefecture. Data sources included the Yunnan Statistical Yearbooks, the Diqing Prefecture Statistics Bureau, the Diqing Tourism Bureau, and the Bank Loan Registration Information System of Diqing Prefecture. Economic data were corrected for inflation using the World Bank's GDP deflator values for China (World Bank 2011).

Results

Multiple forest class change detection

Area-adjusted overall accuracy (i.e., accuracy measures that are corrected for potential bias due to the differences in the proportions of classes in the validation data and the true areal proportions of these classes in the map (Card 1982)) was 92% for the Historic classification and 93% for the Pre-ban and Post-ban classification (Table 3). Old-growth forests were mapped with an accuracy of at least 91% in both classifications. For the change (i.e. deforestation) classes, producer accuracies were lower than the user accuracies, representing a high error of omission, which means that our estimates of forest loss were conservative.

The lowest accuracies were the producer accuracies for the forest cover loss classes during the Historical (78%), Pre-ban (79%) and Post-ban (73%) periods. Accuracies from the raw confusion matrix for these change classes were much higher (ranging from 91 - 94%). However, when adjusting for potential areal biases, accuracies for classes with small areal proportions (e.g. a change class) can diminish dramatically even when just a few points of a smaller class are mis-classified, because these pixels are frequently mis-classified as a class with larger areal proportion (e.g. a no-change class), creating a high error of omission and a low producer accuracy. By generating area estimates for each class, and confidence intervals around those area estimates, a more accurate representation of error was provided (Table 3).

Logging rates and patterns

During the Historic period, 94,763 ± 6,304 ha of forest were logged (95% confidence interval). Logging dropped dramatically during the Pre-ban (58,467 ± 4,645 ha) and Post-ban (57,880 ± 5,202 ha) periods. In the entire study area, the annual disturbance rate decreased from 0.37% in the Historic period, to 0.29% and 0.27% in the Pre- and Post-ban periods, respectively.

The spatial pattern of logging changed over time. Multiple linear regressions to predict logging rates as a function of population and road density were significant during the Historic ($p = 0.04$, Adj. $R^2 = 0.15$) and Pre-ban ($p = 0.01$, Adj. $R^2 = 0.25$) periods, but not significant ($p = 0.59$) during the Post-ban period. In univariate correlations of logging rates with either road or population density, logging rates were positively correlated with road density ($p = 0.05$, $r = 0.36$) during the Pre-ban period, and negatively correlated with village density during both the Historic ($p = 0.01$, $r = -0.46$) and Pre-ban periods ($p = 0.002$, $r = -0.53$). However, during the Post-ban period, there was no significant correlation between logging rates and road density ($p = 0.46$) or logging rates and village density ($p = 0.74$).

Logging rates decreased dramatically throughout the study area after the Historic period, except for the area around Shangrila City, where logging rates were consistently high in all three time periods (Fig. 1e). Furthermore, in Shangrila County, the area of old-growth forest cleared after the ban doubled compared to old-growth forest loss during the Pre-ban period (Fig. 4). Meanwhile, logging of other forest types (pine/oak woodlands and non-pine scrub), and in other counties, remained relatively consistent during the Pre- and Post-ban periods.

Forest cover and regeneration

Total forest cover (old-growth forest, pine/oak woodlands and non-pine scrub combined) increased in the study area from 62% in 1990 to 64% in 2009 (+ 50,000 ha) (Fig. 5a and 5b). This forest area increase represented mainly an increase in the non-pine scrub class (from 11% to 24%), while old-growth forests and pine/oak woodlands declined (from 26% to 20% and from 25% to 20%, respectively). Other land cover types remained fairly constant.

We summarized land cover in 1990-2009 for those pixels that were deforestation or non-pine scrub forest classes in the Historic period to understand trends in the regeneration (Fig. 6). Fourteen percent of the areas logged during the Historic period remained non-forested in 2009 (Fig. 5c). Of the areas where forest cover regenerated, only 14% had a species composition similar to the old-growth forest community, and 8% regenerated as pine/oak woodlands. The remaining 64% of the logged area regenerated as non-pine scrub. Similarly, only a small proportion of the non-pine scrub during the Historic period transitioned to either the old-growth forest community (8%) or pine forest (9%) as of 2009, and a substantial proportion (17%) lacked forest cover (Fig. 5d). The majority of the non-pine scrub in the Historic period remained as such in 2009 (66%). Furthermore, regeneration rates varied spatially. Areas with abundant old-growth forest (Fig. 7a) often had low rates of regeneration of the old-growth forest community (Fig. 7b).

Socioeconomic changes

Economic data indicated that there was rapid development since the logging ban in 1998, due to a rapidly developing tourism industry (Fig. 8a). In 1995, only 40,000 tourists visited Diqing Prefecture, but as of 2009, it received 5.3 million visitors annually. Income from tourism rose from 19 million RMB in 1995 to 5,400 million RMB in 2009. Local government revenue increased 40-fold, from 11 million RMB in 1987 to 440 million RMB in 2009, and annual rural net incomes more than tripled from 661 RMB in 1989 to 2100 RMB in 2009 (Fig. 8b).

Discussion

Multiple forest class change detection

Remote sensing in our study area faces many obstacles, and our study represents the most detailed forest change detection analysis to date for SW China (Willson 2006; Vina et al. 2007; Vina et al. 2008; Tuanmu et al. 2010). Image analysis is challenged by snow cover, strong illumination effects from topography, and senescent vegetation. The collection of ground-truth data is difficult, and forest distribution and composition is extremely heterogeneous. We employed a range of techniques to overcome these obstacles. First, we used SVM to handle complex distributions of our land cover classes. Samples from a wide range of intra-image variability, including dramatic illumination effects and variable snow cover, were included in the training dataset to accommodate the multi-modal and non-normal distributions inherent in such a complex environment. Furthermore, in the field, we specifically identified training data in areas at the spectral boundaries of a class (e.g., 45% pine/oak woodlands and 55% old-growth forest) as input for the SVM. These “mixed pixels” were crucial to aid the SVM to identify precise hyperplanes between the classes.

Second, multi-temporal imagery (Wolter et al. 1995) from late fall and early spring aided the separation of old-growth forest from the pine/oak woodlands and the non-pine scrub classes (Fig. 2). In southwest China, November through April is a period of extended drought, and the fact that different tree species respond differently to drought, was a crucial factor to discriminate forest types. The composite of the multitemporal TM imagery from 1990 with the single winter MSS imagery from 1974 was essential to accurately classify different forest types during the historic period.

Finally, we combined two common approaches to multitemporal change detection -- composite and post-classification change detection techniques – to reduce multiplicative error. The “historical” classification gave us the baseline maps necessary to compare logging rates for different forest types during the Pre- and Post-ban time periods, but we reduced much of the cumulative error that would have resulted from a pure post-classification change detection methodology.

Despite the high level of accuracy achieved by our classifications, there are uncertainties in our estimates. First, it is possible that classification accuracies were overestimated because our ground-truth dataset was not random, possibly resulting in spatial autocorrelation among the pixels within polygons. On the other hand, accuracy was underestimated to some degree because we purposefully included “mixed pixels” in our training and validation dataset to aid SVM in hyperplane placement. These points were often classified inaccurately, even though in the field the appropriate classification is ambiguous.

Second, forests in this region are extremely heterogeneous and we had to limit our classification scheme to just three forest types to achieve a balance between classification accuracy and thematic detail. Third, our classification could not discriminate logging from other forest disturbances (insects and fire), which are not frequent events in our study area, but do occur in isolated patches. In addition, selective logging is not well captured in our land cover classifications. Finally, during post-classification processing we eliminated small, isolated patches to remove noise, and small patches of logging may have been falsely removed. Selective and small-scale logging do occur in our study area, and further research is necessary to quantify the extent and consequences of these logging practices.

Logging Patterns

The logging patterns that emerged from our satellite image analysis reflect NW Yunnan's turbulent history since the mid 1900s, and highlight its consequences for old-growth forest and biodiversity conservation. Our analysis confirmed that logging was indeed intense throughout the study area during the Historic period (1974-1990) (Fig. 1b). Logging rates decreased dramatically during the Pre-ban period (1990-1999), and during the Post-ban period (1999-2009), logging was reduced even further. Thus, our analysis indicates that the bold forest protection policies that China implemented were successful and increased forest cover in our study area.

However, substantial logging still occurred despite the logging ban. Approximately 60,000 ha was logged in each of the Pre-ban and Post-ban periods, and the spatial distribution of logging changed dramatically. We expected that areas with high road density would have high rates of logging (Chomitz & Gray 1996; Cropper et al. 2001). Surprisingly, this was the case during the

Historic and Pre-ban periods, but not in the Post-ban period. Village and road density had some predictive value for logging rates during the Historic and Pre-ban periods, but not during the Post-ban period, indicating that the processes influencing regional-scale patterns of logging changed.

For example, most logging during the Historic and Pre-ban periods was performed by state logging companies. Easily accessible areas (i.e. along major roads) were already logged in high-population density areas, but abundant old-growth forests remained along major roads in low population-density areas, which explains the positive correlation of logging with road density and the negative correlation of logging with village density during these early periods. After the logging ban, state-sponsored logging companies disbanded, and logging was only allowed by local people for non-commercial use. Logging and population/road density became divorced (i.e. relationships became insignificant) during the Post-ban period because logging became concentrated in a “hotspot” around Shangrila.

Old-growth forest logging accelerated in the Post-ban period around the city of Shangrila most likely because of high demand for old-growth timber due to rapid ecotourism-based economic development. Local and national government officials responded to the logging ban by initiating the development of NW Yunnan into one of the premiere ecotourism destinations in China (Morell 2002; Kolas 2008; Jenkins 2009). Shangrila City is the tourism “gateway” into NW Yunnan, and transformed since the 1980s from a backwater town into a one of the premiere tourist destinations in China (Morell 2002; Kolas 2008; Jenkins 2009). The region’s only airport (built in 1999) is located 5 km outside of Shangrila City. In 2007, China’s flagship National Park Pudacuo (Jieng 2008), was established just 30 km from Shangrila City.

The development of Shangrila resulted in a growing demand for tourism accommodations (Fig. 8a) and a growing population (Fig. 8b). From 2002 to 2007, new tourism-based businesses received 2.3 billion RMB in government loans. Shangrila's tourism industry capitalizes on both the natural beauty of the region, and on Tibetan culture (Jenkins 2009). Therefore, much of the new construction for tourists (guesthouses, tourist attractions, restaurants, and vacation homes) uses a Tibetan architecture style (Kolas 2008). In addition, increasing wealth resulted in more - and larger - traditional Tibetan-style houses for local families. Villagers reported to us that each new Tibetan house requires between 50-100 trees, and that the primary sources of timber were the old-growth spruce and fir forests around Shangrila City. Among 23 houses that we visited, those built after 2001 had, on average, central pillars with an average diameter of 79 cm, while pillars in those built before 2001 were only 51 cm in size. These data are too limited to be representative, but indicate potential effects of increasing wealth. In addition to construction materials, each household needs 10-30 m^3 of fuelwood/ year (Xu & Wilkes 2004). Higher demand for firewood due to the increasing number of households, and the restaurants and guesthouses of the tourism industry, thus likely accelerated forest cover loss around Shangrila.

Increased logging rates were likely not a result of increasing demand for wood products in other parts of China, as we did not encounter logging camps or logging trucks, and export of timber to the rest of Yunnan is strictly prohibited and enforced by logging checkpoints. Instead, the increasing demand for old-growth timber in Shangrila is due to the rapidly growing tourism industry and the increasing income levels of the local people. Research in the Wolong Giant Panda Reserve in neighboring Sichuan Province similarly revealed accelerated forest cover loss following reserve implementation, due to tourism, a growing population, and greater wealth (Liu et al. 2001).

Ecotourism, or nature-based tourism, is expanding around the world because it offers a strategy to wed sustainable economic development with environmental protection (Balmford et al. 2009; Karanth & DeFries 2011). However, what constitutes true ecotourism, and the consequences of ecotourism for biodiversity conservation, is still a matter of great debate (Yu et al. 1997; Nash 2009; Kirkby et al. 2010; Sims 2010; Wang & Buckley 2010). In Shangrila, the tourism industry encompasses a wide range, including nature, adventure, ethnic, and protected area tourism. We used the term “ecotourism” to represent this wide range because, first, appreciation of nature is undeniably the primary attraction of Shangrila. Second, and perhaps more importantly, Shangrila’s tourism industry is branded as ecotourism, i.e., both the producers (advertisers, local businesses, tour guides) and consumers (i.e. tourists) believe that they engage in ecotourism. As such, it is particularly noteworthy that our result showed continued deforestation.

Forest cover trends

While old-growth forests declined, forest area overall increased from 62% in 1990 to 64% 20 years later. However, only a fraction (20%) of deforested or non-pine scrub areas during the Historic period returned to either pine/oak woodlands or the old-growth forest vegetation community (Fig. 5c and d). This is not to say that with time and appropriate forestry management, the non-pine scrub forests in our study area could not potentially return to the original vegetation community, but we did not observe this. No information is available about successional trajectories of Shangrila forests, but in general, regeneration to climax forest in temperate regions takes centuries to thousands of years, much slower than the observed rate of old-growth forest clearing around Shangrila. Furthermore, our field observations indicated that land use intensity was high in regenerating areas, inhibiting the regeneration of the original vegetation community. Likewise, our remote sensing results suggest that non-pine scrub may not be in a process of natural succession, since we observed small proportions of regeneration to pine or vegetation communities typical of old-growth forests in those areas that were non-pine scrub forest from 1974-1990.

The assumption of forest transition theory is that increasing forest cover indicates that environmental conditions are improving (Mather et al. 1999), but here we show that increasing forest cover alone does not necessarily mean that biodiversity and natural ecosystems are on a pathway towards recovery (Meyfroidt & Lambin 2011). We observed that secondary and scrub forests are a growing proportion of forests in our study area, which is similar to many countries around the world (Chazdon 2008; Grau et al. 2008). Yet, most studies showing shifts in forest trajectory have not distinguished between primary and secondary forest types, even though the value of regrowing forests, in terms of species habitat and ecosystem services, is far lower than that of old-growth forests (Chazdon 2008; Meyfroidt & Lambin 2008; Perfecto & Vandermeer 2010). Land use largely determines whether the forests regrowing after harvest will eventually recover to a high-diversity ecosystem (Chazdon et al. 2009), but unfortunately, land use intensity in secondary forests is not included in forest transition theory (Meyfroidt et al. 2010; Perfecto & Vandermeer 2010). Likewise, forest transitions and increasing forest cover may be only due to monoculture plantations with little biodiversity value (Rudel 2009) or result in more intense land use of surrounding forests, with potentially negative effects on overall biodiversity (Perfecto & Vandermeer 2010). Furthermore, countries experiencing a forest transition may simply export their ecological footprint elsewhere (Mayer et al. 2005; Meyfroidt & Lambin 2009; Meyfroidt et al. 2010).

Conclusions

Our results highlight that forest monitoring must incorporate multiple forest classes to assess forest change in the context of the conservation of biodiversity and ecosystem services. Simple forest versus non-forest cover assessments in areas with remaining unprotected old-growth forests are inadequate to understand the implications of protection and development strategies for high-diversity forest types, and can obscure important environmental degradation processes. This represents a major challenge for the remote sensing community, because error and complexity increases when multiple dates or multiple classes are examined via change detection. We used multi-temporal imagery, carefully selected training data with SVM and a combination post-classification and composite classification technique to reduce error and accurately detect change in multiple forest classes in a complex environment.

One of the primary objectives of the NFPP is to ban commercial logging of all forests in southwest China, and allow only small quotas for local consumption. Our results show that, overall, China's forest protection policies effectively reduced forest loss in NW Yunnan. Logging decreased over most of the landscape, and forest cover increased. However, the logging ban was trumped in areas of rapid economic development, as old-growth forest loss accelerated due to a growing ecotourism industry. Ecotourism has expanded rapidly in developing countries around the world (Balmford et al. 2009; Karanth & DeFries 2011), because it offers a strategy to wed sustainable economic development with environmental protection. However, our results show that even in Shangrila – an arguably best-case-scenario with strong institutions, well-funded environmental protection efforts, and strong government policies aimed at forest protection -- the negative impacts of ecotourism-based economic development on the environment outweighed conservation efforts. As tourism development continues to expand into previously remote and little-visited regions, the negative impacts observed near Shangrila City in the last decade may soon follow unless steps are taken to mitigate the threats that development poses.

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Table 1. Images used for the analysis.

	Time Period						
	ca. 1974	ca. 1990		ca. 1999		ca. 2009	Reference
Landsat Sensor	MSS	TM	TM	TM	ETM	TM	ETM
Acquisition Date	1/5/1974	11/20/1990	4/13/1991	10/28/1999	4/13/2000	11/24/2009	12/25/2000

Table 2. The number of ground truth polygons and pixels in each class.

Land Cover Class	**Polygons**	**Pixels**
Old-growth forest	396	18664
Pine/oak woodland	184	9713
Non-pine scrub	329	24149
Deforestation - historic	51	5442
Deforestation pre-ban	85	3435
Deforestation post-ban	81	3970
Agriculture	141	5710
Grassland	98	2244
Alpine shrub expansion	41	1826
Sparse shrub	78	12198
Bare/urban	45	2063
Other	44	3886
Total	1573	93300

Table 3. Producer (PA) and user (UA) accuracies, adjusted areal extent and confidence intervals (CI) for each land cover class.

HISTORIC CLASSIFICATION

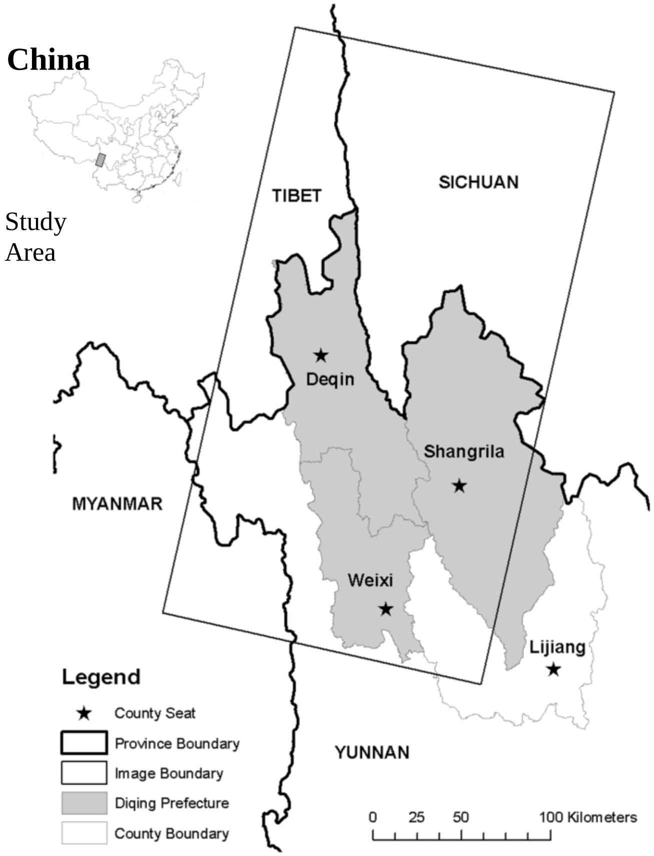
Class	PA	UA	Adj. Area (ha)	±CI (ha)	±CI (%)
Old-growth forest	93%	95%	566233	11976	2%
Pine/oak woodland	93%	89%	513940	13562	3%
Non-pine scrub	86%	86%	236890	10001	4%
Deforestation Historical	78%	95%	94763	6304	7%
Agriculture and Grassland	89%	94%	226072	7527	3%
Other	95%	93%	500732	10247	2%
Overall accuracy =	92%				
Khat =	0.92				

PRE-BAN AND POST-BAN CLASSIFICATION

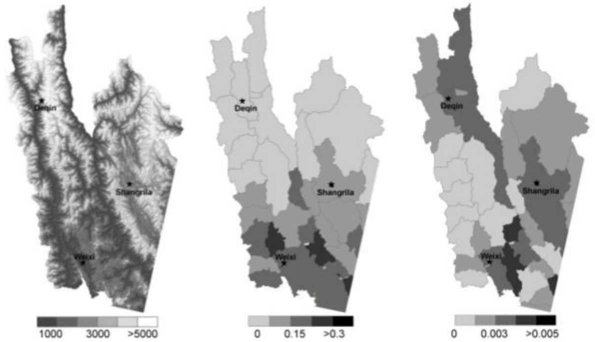
Class	PA	UA	Adj. Area (ha)	±CI (ha)	±CI (%)
Old-growth forest	91%	91%	501165	12435	2%
Pine/oak woodland	93%	92%	517316	11836	2%
Non-pine scrub	97%	90%	533596	11696	2%
Deforestation Pre-ban	79%	96%	58467	4645	8%
Deforestation Post-ban	73%	95%	57880	5202	9%
Agriculture	97%	98%	138090	2600	2%
Grassland	92%	99%	68710	3201	5%
Alpine Shrub Expansion	93%	99%	79476	3153	4%
Sparse Shrub	95%	97%	96842	2850	3%
Bare/Urban	86%	95%	106076	5215	5%
Other	97%	96%	392733	6097	2%
Overall accuracy =	93%				
Khat =	0.93				

Figure 1. a) Location of Diqing Prefecture, in Northwest Yunnan, China, b) elevation, c) population density, d) road density, and e) annual disturbance rates by township for the Historic (1974-1990), Pre-ban (1990-1999) and Post-ban (1999-2009) periods.

a) Study Site Location

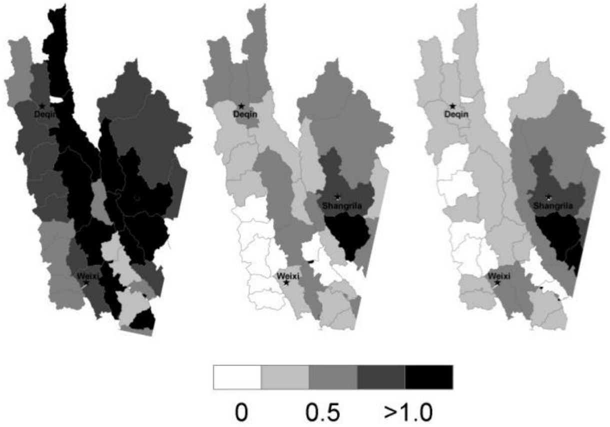


b) Elevation (m) c) Population density (villages/km^2) d) Road density (km of road/km^2)



e) **ANNUAL FOREST DISTURBANCE RATE**

Historic Pre-ban Post-ban



H

Figure 2. Examples of how phenology was used to discriminate the different forest classes. Representative pixels from the three forest classes look similar under visual inspection on Landsat images from A) November 20, 1990 and B) April 13, 1991, but C) spectral plots (Landsat bands 1 through 6) show that different forest types vary in their response to winter drought, enabling discrimination between forest types with multi-temporal imagery.

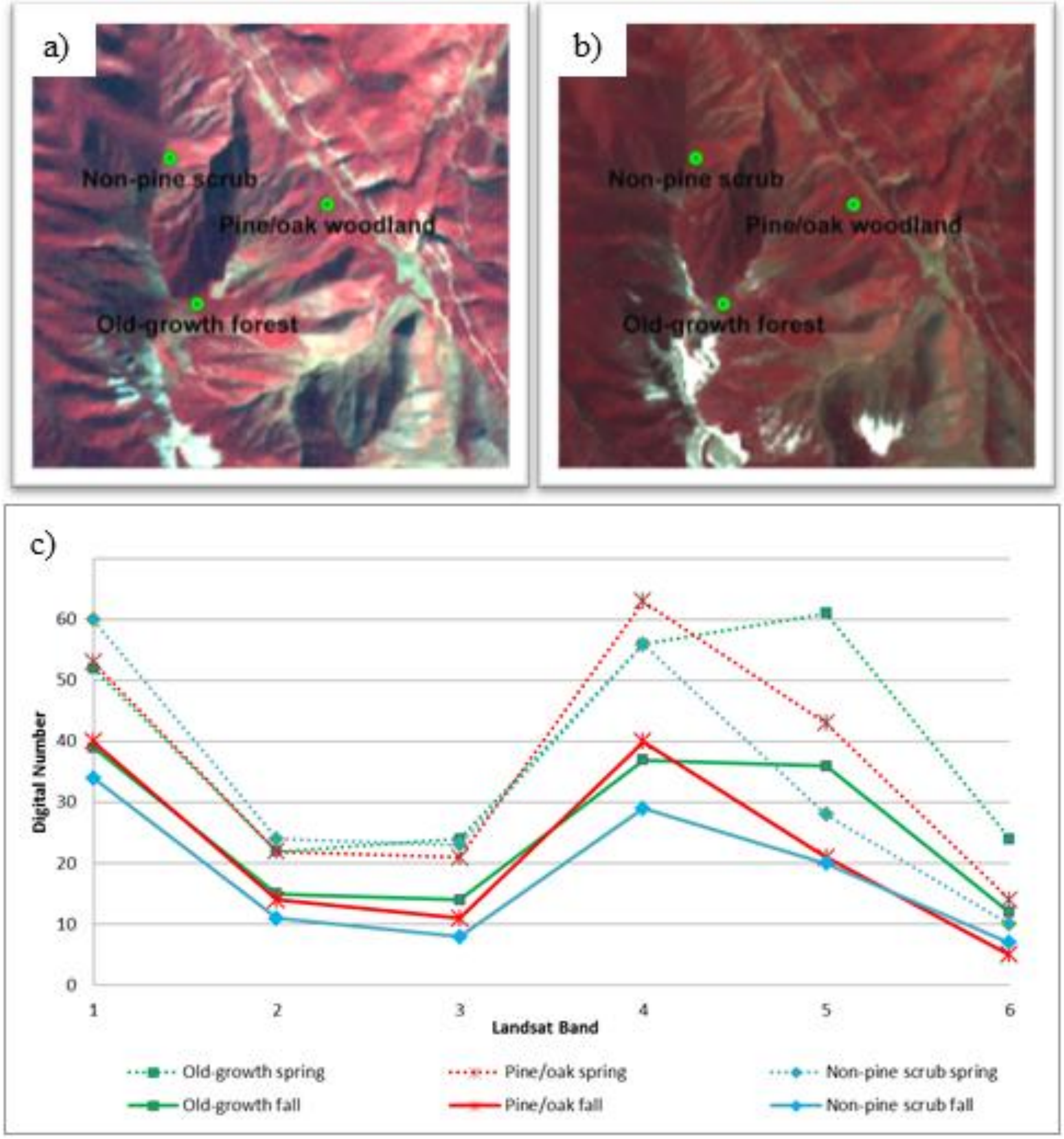


Figure 3. Description of how multitemporal imagery was used in a combined composite and post-classification change detection approach.

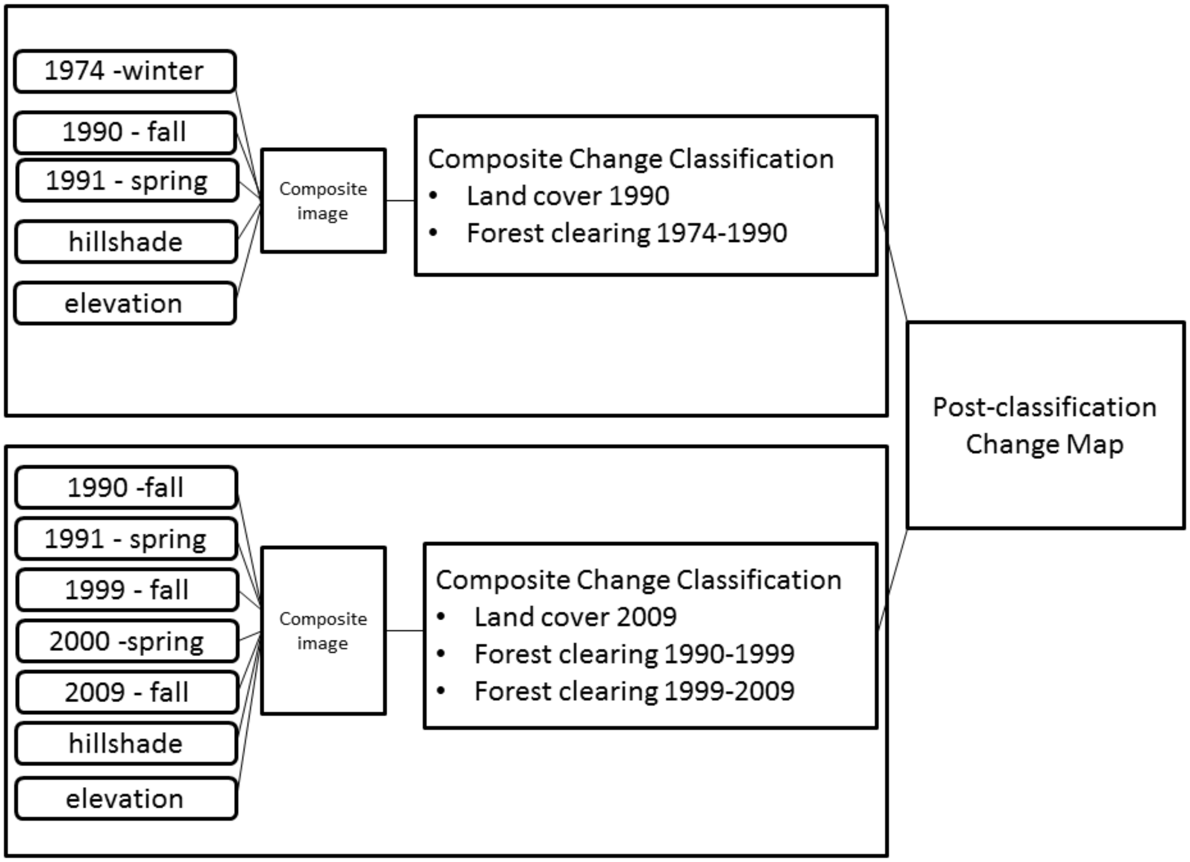


Figure 4. Hectares of forest loss by forest type during the Pre-ban (1990-1999) and Post-ban (1999-2009) periods.

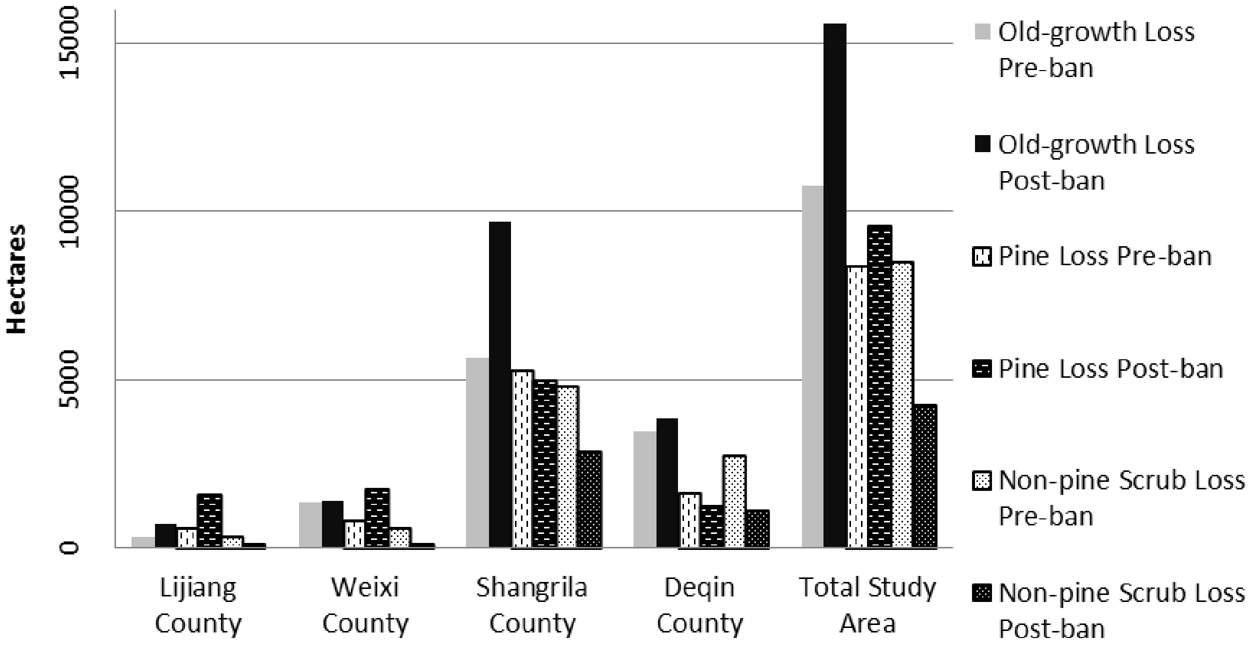


Figure 5. Land cover distributions in a) 2009 and b) 1990 for the entire study area. The 2009 land cover of areas c) logged during the Historic period (1974-1990) and d) classified as secondary forest/shrub during the Historic period.

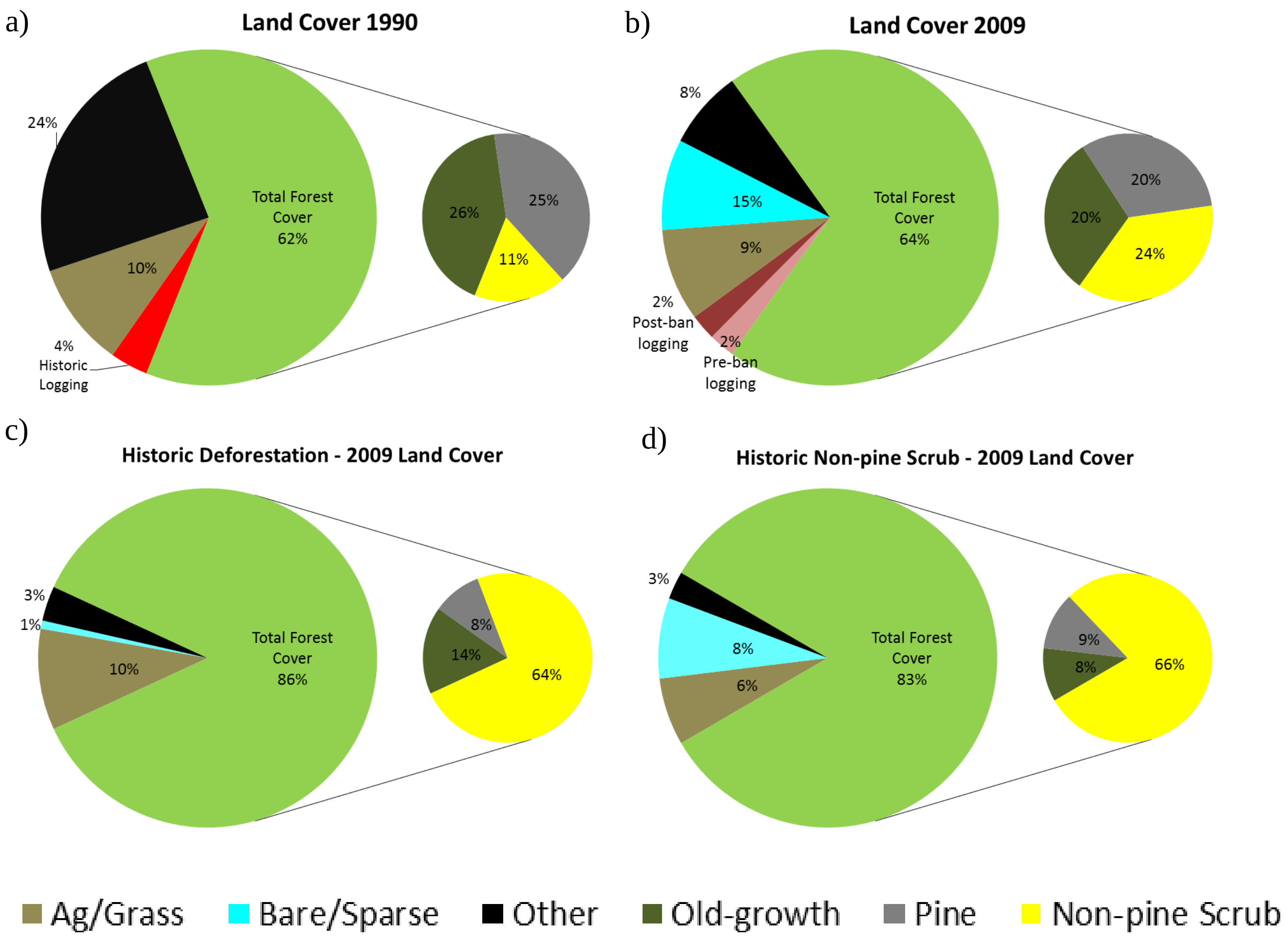
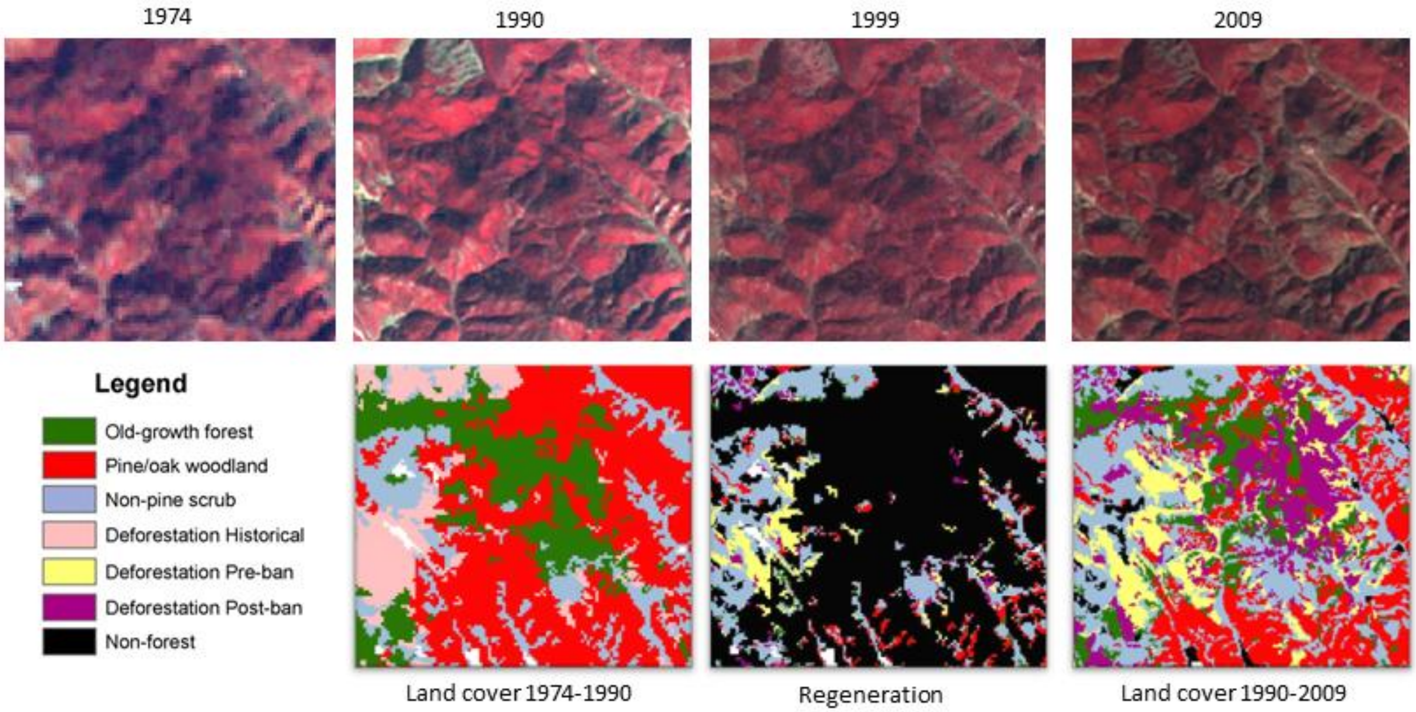


Figure 6. A subset of the classifications for an area 8 km north of Shangrila that experienced intense logging during the Historic, Pre-ban and Post-ban periods. a) Logging activity and non-pine scrub surrounded a large patch of old-growth and pine forests during the Historic period. b) The composite classification from 1990-2009 for only those areas that were logged or non-pine scrub during the Historic period show that the majority of these areas regenerated as non-pine scrub. Exceptions include regenerating pine plantations and sites that continued to be logged. c) During the Pre-ban and Post-ban periods, the large patch of pine and old-growth forest from (a) experienced intense logging.

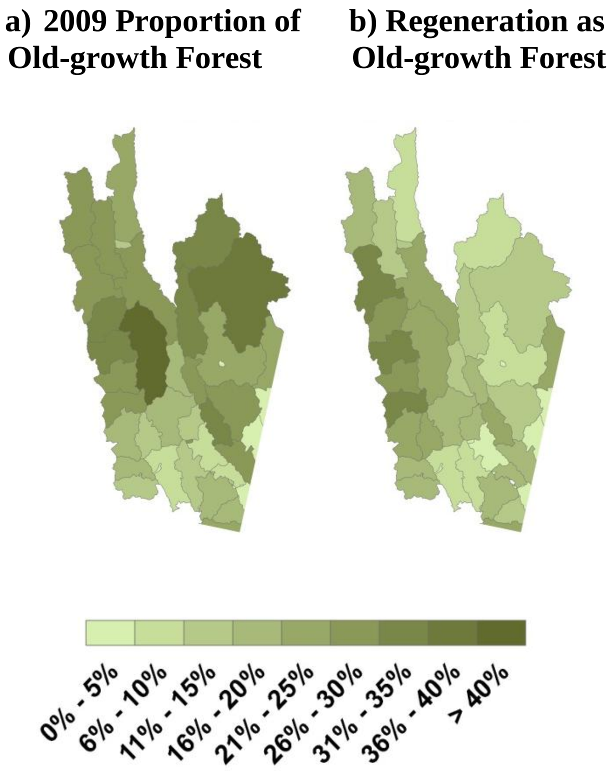


Land cover 1974-1990

Regeneration

Land cover 1990-2009

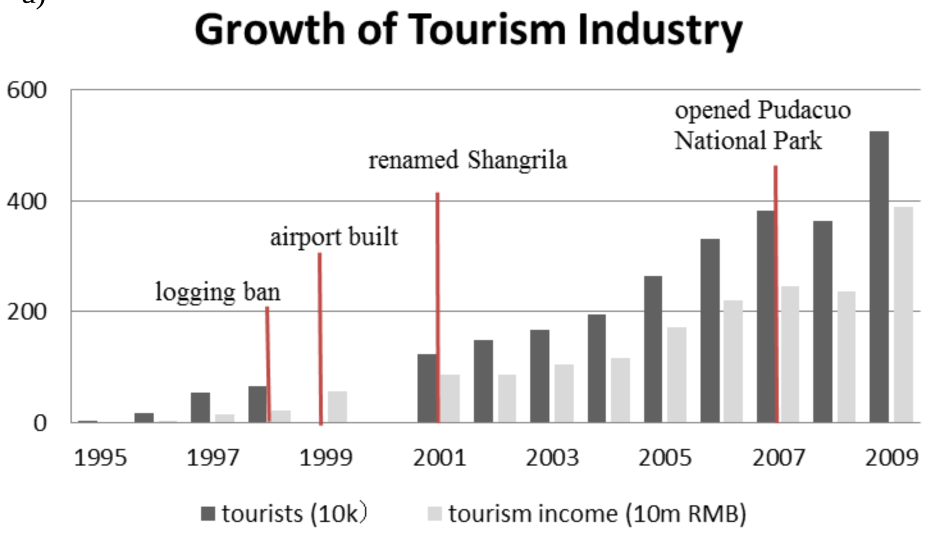
Figure 7. Patterns of (a) current proportion of old-growth forest on the landscape and (b) relative rates of regeneration as old-growth forest.


a) 2009 Proportion of Old-growth Forest

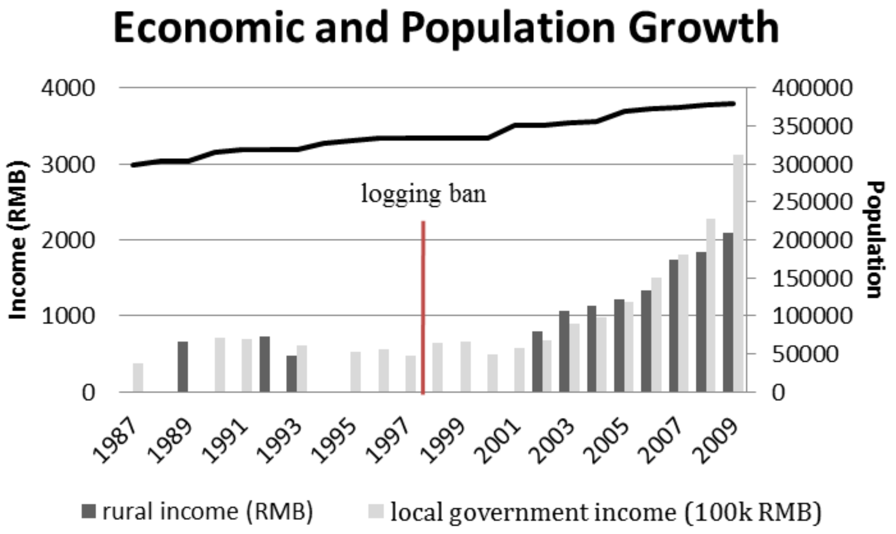
b) Regeneration as Old-growth Forest

Figure 8. a) Economic development and population growth, and b) growth of the tourism industry in Diqing Prefecture.

a)



b)



Chapter 2: Regime shift on the roof of the world: Alpine meadows converting to shrublands in the southern Himalayas

Abstract

Worldwide, changing climates and land use practices are escalating woody-plants encroachment into grasslands, reducing biodiversity and altering ecosystem functions. The loss of alpine grasslands is a major conservation concern as they harbor many rare and endemic species. Alpine meadows in Northwest Yunnan, China, represent a global biodiversity hotspot with high species richness, beta diversity, and endemism. Shrubs have expanded greatly in the region and threaten alpine meadow biodiversity. To measure rates of meadow loss due to shrub encroachment and identify its mechanisms, we reconstructed alpine land cover, climate, and land use change from 1950 to 2009 across Northwest Yunnan using satellite data, ground surveys, and interviews. Between 1990 and 2009, at least 39% of the alpine meadows converted to woody shrubs. The patterns of change suggest that a regime shift is occurring. Despite multiple perturbations to the climate and land use systems starting in the 1950s, alpine meadows remained resilient to shrub expansion until the late 1980s. Shrublands rapidly expanded then due to feedback mechanisms involving climate, woody cover, and grazing. Fire is no longer an effective tool for controlling shrub expansion. This regime shift threatens both endemic meadow biodiversity and local livelihoods. More generally, these trends serve as a warning sign for the greater Himalayan region where similar vegetation changes could greatly affect livelihoods, hydrology, and climate.

Introduction

Worldwide, shrubs are encroaching in grasslands, affecting biodiversity, ecosystem function, and carbon storage (Knapp et al. 2008). Climate change, overgrazing, and land use all contribute to shrub encroachment (Walker & Wahren 2006). When multiple interacting factors reach a critical threshold, a relatively small perturbation can trigger a regime shift, i.e., a sudden, and potentially irreversible, transition to an alternate state (Scheffer 2009). If we want to conserve grassland biodiversity, then we need to identify the drivers of shrub encroachment and their critical thresholds. Untangling these factors, however, is challenging in coupled natural and human systems (Brock & Carpenter 2010).

Shrub encroachment into alpine meadows represents a particular grave conservation threat (Xu & Wilkes 2004). Alpine meadows often harbor many endemic plants adapted to harsh conditions, and orogenies have resulted in high levels of both paleo- and neo-endemism (Wu et al. 2007). With the largest alpine ecosystems in the world, alpine meadows in the Greater Himalayan region are of particular concern. Northwest Yunnan Province in southwest China represents a global biodiversity hotspot (Myers et al. 2000), and meadows here have higher species richness, beta diversity, and endemism than elsewhere in the Himalayas (Salick et al. 2009; Xu et al. 2009).

Historically, Himalayan meadows have been used by indigenous agro-pastoralists whose rangeland management practices sustained both local livelihoods and biodiversity (Klein et al. 2011). The highly nutritious meadows are critical to Tibetan transhumance herding, and the replacement of meadows by unpalatable shrubs threatens their livelihoods (Yi et al. 2007). Alpine meadow species can persist in the low-intensity land use systems typical for montane regions, but even subtle environmental changes may threaten them (Thuiller et al. 2005; Galop et al. 2011). For example, when climate change shifts woody vegetation to higher elevations, species of alpine mountaintop meadows may not be able to migrate to other suitable environments (Parmesan 2006). Likewise, the abandonment of traditional land use practices can result in shrub encroachment (Laiolo et al. 2004; Baur et al. 2006). Indeed, climate change and land use change interact in alpine ecosystems making it difficult to identify the main driver(s) of shrub expansion.

Since the 1950s, numerous environmental changes affected Himalayan ecosystems. Climate warming rates here are higher than the global average, changing ecosystem productivity (Peng et al. 2010), phenology (Yu et al. 2010a), and permafrost (Xu et al. 2009). Changing political and social structures have led to economic development, and abandonment of traditional land use practices (Yeh & Gaerrang 2010; Klein et al. 2011). However, despite grassland degradation throughout the Himalayas (Harris 2010), rates of shrub encroachment have not been quantified, and its drivers are not well understood.

Our goals were a) to measure rates of shrub expansion into alpine meadows in Northwest Yunnan and b) to identify the likely mechanisms driving these changes. We used Landsat TM/ETM+ satellite image analysis and dendrochronology to map land-cover distribution and change in the forest/shrub/meadow ecotone of the alpine zone (3,800 - 4,500 m) in 1974, 1990, and 2009. In addition, we reconstructed changes since the 1950s using interviews, livestock records, and climate data. We then tested the following hypotheses:

H1. *Climate change*: We hypothesized that temporal patterns of shrub establishment and expansion were caused by changes in temperature and precipitation. Warming alters phenology (Sturm et al. 2005; Yu et al. 2010a) and caused shrub encroachment in other alpine (Harte & Shaw 1995; Klein et al. 2007) and arctic (Molau 2010) environments. Snowpack variability can alter productivity of high-elevation systems (Peng et al. 2010) and cause shrub encroachment (Bjork & Molau 2007; Wipf & Rixen 2010). In the Himalayas, temperature, precipitation, snowpack, and phenology changed rapidly since the 1950s (Xu et al. 2009), causing glaciers to recede and tree-lines to rise (Baker & Moseley 2007b; Wong et al. 2010).

H2. *Burning cessation*: We hypothesized that the 1988 region-wide burning ban triggered shrub establishment, and that areas where burning continued irrespectively suffered less shrub encroachment. Burning cessation has triggered shrub encroachment world-wide (Knapp et al. 2008). In NW Yunnan, herders historically burned shrubs to improve meadows, but the 1988 burning ban may have fostered shrub encroachment (Sherman et al. 2008).

H3. *Shrub autocatalysis*: We hypothesized that shrub autocatalysis, where existing shrubs facilitate further shrub expansion due to feedbacks (Sturm et al. 2005), was an important driver of shrub expansion. Feedbacks are critical components of regime shifts, because they push the original ecosystem towards an alternate stable state from which reversion to the original state is difficult or impossible (Scheffer 2009).

H4. *Overgrazing*: We hypothesized that areas of high grazing intensity had more shrub encroachment, because shrubs establish more easily when other vegetation is removed (Xu & Wilkes 2004; Roder et al. 2007; Galop et al. 2011).

H5. *Abandonment*: We hypothesized that territories where alpine yak herding was abandoned have particularly high shrub encroachment rates. Abandonment was the major driver of 20th century shrub encroachment in the European Alps (Galop et al. 2011). In our study area, economic development integrated rural areas into the regional economy of western China since the 1980s (Yeh & Gaerrang 2010), leading to abandonment of yak herding in some villages (Yi et al. 2007).

Materials and methods

Study area

Northwest Yunnan Province is at the southeast edge of the Qinghai-Tibetan Plateau (QTP), bordering Tibet and Sichuan Province (Fig. 9), and an IUCN biodiversity hotspot (Myers et al. 2000). Three major rivers (the Yangtze, Mekong, and Salween) create steep gorges, with elevations ranging from 1,800 to 6,740 m. The alpine zone occurs between 4000 and 5000 m. Climate in the alpine is cold with near-constant rain during the monsoon season (June-September) and sporadic snowfall from October through May. The mean annual temperature above 4,100 m is < 0° C (Sherman et al. 2008).

The alpine zone consists of shrub, meadow, and scree ecosystems. Meadows exhibit the highest species richness and support the most rare and endemic plant species (Sherman et al. 2008; Salick et al. 2009). Alpine meadow communities are dominated by Polygonum viviparum, Potentilla fulgens, Fragaria nilgeerensis, Anemone narcissiflorioides, Cyananthus flavus, and Festuca ovina and include Pedicularis dolichoglossa, Viola szetchuanensis, Agrostis limprichtii, and Primula sp. (Li & Walker 1986a), and beta diversity is high (Sherman et al. 2008; Salick et al. 2009).

Alpine meadows are crucial to local Tibetans' yak husbandry because meadows provide the highest nutritional value (Yi et al. 2007). When interviewed, herders confirmed the increase in shrubs, and cited it as a major threat to livelihoods (Haynes 2011). The shrub invaders include

Rhododendron spp., Juniperus spp., Berberis spp., Potentilla fruticosa and P. glabra, all of which are either noxious or poor forage for grazers.

Satellite Image Analysis

The study area is covered by two Landsat footprints (path/row 132/040 and 132/041). We acquired Landsat MSS images from 1974, and Landsat TM/ETM+ images from circa 1990, 2000, and 2009, and co-registered all images to the orthorectified Landsat 2000 GeoCover Dataset (Brandt et al. in review). Positional uncertainty was < 0.4 pixels (<12 m). Clouds and cloud shadows were masked manually.

We collected field data to train and validate the satellite classifications in July-November 2008, August-October 2009, July-September 2010, and May-July 2011. Due to the ruggedness of the study area, a random sampling design was not feasible. We collected field data in nine alpine territories prior to image analysis, and nine different alpine territories thereafter (Table 4). We hired local guides, conducted informal interviews about land-use, and recorded land use for at least 40 locations within each land cover type and took photographs of the surrounding 1 ha (Justice & Townshend 1981). In addition, we gathered ground-truth data from high resolution imagery in GoogleEarth^{TM} (approximately 20% of the ground truth dataset). In total, our ground truth dataset included 1,573 polygons (93,300 pixels).

To classify land cover change, we used Support Vector Machines (SVM). We assessed two time periods, 1974-1990 (historic period) and 1990-2009 (recent period). For the historic period, we combined 1974 MSS images, circa 1990 TM images, and elevation and hillshade (corresponding to the illumination of the November 19, 1990 Landsat TM image), and carried out a composite analyses (Kuemmerle et al. 2006; Brandt et al. in review) separating four classes: Grassland, Shrubland, Forest, and Snow. Similarly, we combined TM/ETM+ imagery from 1990, 2000 and 2009, along with elevation and hillshade data (October 28, 1999 image) to classify the same land covers, plus the change class “shrub encroachment”, i.e., those areas that converted from grassland in 1990 to shrubland in 2009. To fill cloud and snow gaps, we carried out a classification excluding the spring images (where snow is more extensive). To train and validate our SVM, we randomly selected approximately 1000 points from each class in our ground truth dataset and used ten-fold cross-validation (Brandt et al. in review). We used a minimum mapping unit of 6 pixels (~ 0.65 ha).

To validate our change maps, we calculated a confusion matrix, overall, user’s and producer’s accuracies, plus area-adjusted estimates of land cover proportions and their confidence intervals (Card 1982). For the accuracy assessment, we used a total of 231 independent ground truth points, including 153 field points from 11 post-hoc field visits in 2010 and 2011. We also randomly sampled 78 points using GoogleEarth high-resolution imagery. For the shrub encroachment class, we relied solely on field data. Acquiring independent data for the historic shrub encroachment class was not possible retrospectively and we report the cross-validation accuracies only.

We estimated shrub encroachment first as the encroachment rate (ER), i.e., the proportional (%) area of grassland in 1990 that converted to shrubland by 2009.

$$ER = 100 * E/(E + PG) \quad \text{Eq. 1}$$

Here, E is the number of pixels in the encroachment class, and PG is the number of pixels in the Permanent Grassland class. This measure is a slight overestimate, because most still snow-covered areas in the imagery were grasslands. Therefore, we also calculated a snow adjusted encroachment rate (ERA) which assumes that all of the snow-covered areas were grassland:

$$ERA = 100 * E/(E + PG+S) \qquad Eq.\ 2$$

where S is the number of snowy pixels. This represents a conservative rate of shrub encroachment, because a small percentage of the snow-covered areas are shrub, bare ground, scree, or bedrock.

We summarized encroachment rates and land cover for 1990-2009 at several different spatial scales. First, we summarized alpine land cover and change for the entire study area, and inside and outside of the Baimaxueshan National nature reserve, where burning prohibitions are most strictly enforced. Second, we summarized land covers for zones of 100-m elevation, 5^{o} slope, and 45° aspect. Third, we summarized land cover change at the township level. Finally, we summarized encroachment and land cover within the alpine “territory” of each village where we had conducted interviews with herders (see below).

Dendrochronology

We took dendrochronology samples from 125 shrubs in 13 sites and five territories. Prior to classification, we collected these shrub cuttings in areas of mixed meadow and shrub land cover, for use in training the image classification. Following image analysis, we sampled areas mapped as encroachment. At each site, we cut discs from the bases of at least 3 of the largest individuals of the dominant shrub species. We dried and sanded the discs, and counted rings using a hand lens (Liang & Eckstein 2009).

Formal Interviews and Livestock Records

We conducted semi-structured interviews with 37 herders in six villages during the winter of 2010. The survey instrument contained seven sections: background of the interviewee, animal husbandry, burn history, village history, livestock health, ecology, and future expectations (for details see Haynes 2011). Interviews were conducted in the local version of Mandarin Chinese (known as Zhongdian hua) or in the local Tibetan dialect and interpreted into Zhongdian hua by the local village host. We collected data on general burning practices in five time periods: feudal (prior to 1958), collectivization (1959-1982), decollectivization (1983-1987), post burning restrictions (1988-1999), and present time (2000-2010). We calculated the percent of respondents who participated in, or had heard of, burning during each of these five time periods. We also interviewed officials from the grassland monitoring station and animal husbandry bureau.

In addition to the formal interviews, we conducted informal interviews with our guides about land-use in 15 different alpine territories while collecting ground-truth data. Livestock data were collected from published literature, including the Deqin County and Diqing Prefecture Almanacs, and annual statistical almanacs (Deqin 1997; Le'anwangdui 2001).

Climate Data

We analyzed meteorological station data maintained by the National Meteorological Administration of China from the two meteorology stations in our study area, Zhongdian and Deqin counties, and calculated local variation in the rates and seasonality of change over the last half century (1960-2008). We used the cumulative method (Lozowski et al. 1989) to filter out noise. We calculated 24- and 48-year rates of change for precipitation, annual daily low, annual daily high, growing season daily low, growing season daily high, winter daily low, and winter daily high (Haynes 2011).

Alpine vegetation is vulnerable to snowpack and snowfall variability. Ideally, our climate data would have included long-term data on snowpack and snowfall change in the alpine zone. Lacking such data, we estimated alpine zone temperature changes from 1960 to 2009. We were especially interested in periods when average daily low temperatures changed from sub-freezing to above-freezing temperatures.

To estimate temperature changes at alpine elevations, we used 48-year temperature data at the Deqin County station (3320 m), and data from 13 stations ranging from 1646 to 4292 m within Deqin County based on at least 20-year means (Le'anwangdui 2001). We calculated the linear lapse rate adjustment for each month (Dodson & Marks 1997) since this technique provides more detail than the global mean lapse rate of -6.5°C km^{-1} (Barry & Chorley 1987), does not require interpolation which is often inaccurate for the Himalayas (Leemans & Cramer 1991; Rolland 2003), and was spatially detailed enough for our analysis. Average daily low and high temperatures, as recorded at Deqin climate station, were then used to calculate low and high temperatures at 3800, 4000, 4200 and 4400 m using the lapse rate adjustments for February through June.

Results

Changes in Alpine Land Cover

Land use and land cover in the alpine zone (Fig. 9b) were mapped with an overall accuracy of 94% and 93% in the recent and historic time period, respectively (Table 5). The overall accuracy of the shrub encroachment class was 91%, and the minimum accuracy of any class was 80%.

Changes in the proportions of grass and shrub between 1974 and 1990 were negligible, and forest cover remained largely unchanged from 1974 to 2009 (Fig. 9c). However, from 1990 to 2009, meadows shrank from 20% to 7%, representing a shrub encroachment rate of 65%. Even the conservative adjusted encroachment rate (which assumes snow-covered areas to be unchanged grassland) was 39%. All rates reported in the following are adjusted encroachment rates.

Spatial patterns of Shrub Encroachment

Shrub encroachment rates varied across the study area, with low rates in the western and southern townships (~5%) and high rates in the northern and eastern townships (>65%) (Fig. 10a). Encroachment rates across the study area were negatively associated with the proportion of spring snow cover from the image classification (Pearson's r = -0.687) and positively correlated with the proportion of woody cover in 1990 (Pearson's r = 0.692, Fig. 10b).

Meadows at the lower elevations of the alpine zone (between 4000 m and 4200 m) were encroached most rapidly (>40%), whereas encroachment rates in the sub-alpine (3800-4000m) and upper alpine meadows (4300-4500m) were lower (Fig. 10c). Meadows on the sunnier south, SE, and SW facing slopes were encroached twice as fast (40-50%) as those on north-facing slopes (18-23%) (Fig. 10d). Encroachment rates did not vary among different slope zones.

Encroachment rates near the national highway were highest (~60%) and declined further away (to about 40%) (Fig. 11a). Encroachment rates within the Baima Reserve (26%), where the burning ban was strictly enforced, were lower than outside the reserve (42%) (Fig. 11b). Remote villages where patchy burning was still practiced had some of the highest rates of encroachment (>50%) compared to <41% in villages where burning was no longer practiced. The two territories that were abandoned exhibited both one of the lowest (12%) and the highest (72%) encroachment rates (Fig. 11b).

Temporal trends in Shrub Establishment

According to the satellite classification, eight of the 13 sites sampled for the dendrochronology analysis converted from grass to shrub from 1990 to 2009, three sites were stable grasslands, and two sites were already shrub-dominated before 1990 (Table 6). Seven of the eight converted sites had > 50% shrub cover according to field visits. All converted sites contained at least three different shrub genera, but dwarf rhododendron was dominant in seven of the eight converted sites. The oldest shrub in each site (typically *Juniperus spp.*) established around 1980 (age = 29 years (SD 4.2)) and the oldest rhododendron shrubs established in 1988 (age = 21 years (SD 8.1)).

Changes in Land-use Practices

Our interviews and official livestock records highlighted that alpine herding practices varied greatly as national policy shifted (Yeh & Gaerrang 2010). During the Feudal era (pre-1958), feudal chiefs or monasteries owned and managed the livestock, and yak herds were small (< 50,000 head in 1952). Grasslands were abundant and burning was limited (only 11% of respondents reported burning). During the Collectivization era (1958-1983), livestock and land were collectivized and there were high incentives to increase production. Herd sizes grew to >150,000 yaks by 1983, and alpine burning was at its peak (54% of respondents reported burning). Following decollectivization (1983-present), families for the first time owned and managed livestock and land, and burning diminished (to 40%). Burning decreased further after the 1998 burning restrictions (to 22% from 1988 to 1999) and then to only 11% from 2000 to 2009). Yak herds grew to a peak of almost 250,000 in 2001.

Climate Change

Deqin and Shangri-la counties experienced substantial changes in temperature and precipitation from 1960 to 2008, representing a hotspot of change even within the Greater Himalayas (Haynes 2011), which is warming much faster than the global average (Yu et al. 2010a). At the Deqin County station the rate of daily high temperature change over the past 24 years was 7.2°C/100 years and the rate of change in daily low temperatures was 1.4°C/100 years. Precipitation changes were especially pronounced in spring (February through May) after 1980. The overall amount of precipitation during these months did not change considerably, but there was no snowfall (Fig. 12a) and increased rainfall (Fig. 12b) after 1980 at 3,320 m.

At higher elevations, average estimated daily low and high temperatures generally increased from 1960 to 2008 (Fig. 12c and 12d). The most dynamic oscillation around the 0^{o} mark occurred in May between 1980 and 1990, with minimum temperatures at 4000 m rising consistently above freezing by the mid 1980s. The linear lapse rates fit well within global estimates and followed the expected trend from months of lower moisture to higher moisture (Barry & Chorley 1987; Dodson & Marks 1997; Rolland 2003) (February, -6.65°C km^{-1}; March, -6.35°C km^{-1}; April, -6.25°C km^{-1}; May, -5.91°C km^{-1}; June, -5.59°C km^{-1}, all $r^2 > 0.97$).

Discussion

Implications of Shrub Encroachment for Biodiversity

Alpine meadows remained unchanged from 1974 to1990 despite changes in climate and land use, but an abrupt shift occurred from 1990 to 2009 when at least 39% of all meadows were encroached by shrubs. This represents a very rapid encroachment rate encroachment (2.1% per year) higher than rates reported elsewhere (Briggs et al. 2005; Coop & Givnish 2007; Thompson 2007; Sankey & Germino 2008). Given the exceptionally rich flora with many endemics, this shrub encroachment is likely to seriously deplete plant diversity in the region (MacArthur & Wilson 1967) and represents a major conservation threat.

Testing Hypotheses

Based on the patterns of shrub encroachment, and climate change (Fig. 13), we tested our hypotheses about potential drivers of shrub encroachment.

H1. *Climate change*: The rapidly increasing temperatures since the 1970s in general, and the sharp reductions in spring snow cover in particular, likely contributed to shrub encroachment. Across the region, areas that retained spring snow cover had low rates of shrub encroachment (Fig. 10b). Decreasing snow cover gives a competitive advantage to shrubs over herbaceous species in alpine and arctic environments (Bjork & Molau 2007; Wipf & Rixen 2010). Our alpine temperature estimates indicate that the most dynamic temperature changes occurred during late spring (May) in the mid 1980s when average daily low temperatures rose above freezing (Fig. 13). This change may have triggered the initial establishment of dwarf rhododendrons in 1988.

H2. *Burning cessation*: Contrary to our expectations, rates of shrub expansion were higher in areas where burning continued relative to those that did not burn (Fig. 12b). Thus, restrictions on burning do not appear to have driven shrub encroachment. However, the region-wide burning ban in 1988 may have fostered initial establishment of dwarf rhododendrons that then spread rapidly.

H3: *Shrub autocatalysis*: Shrub encroachment rates were higher in areas where woody vegetation was already prevalent (Fig. 10b), suggesting that shrub autocatalysis fostered shrub encroachment. Shrub autocatalysis is a result of positive feedbacks, and common in shrub expansions worldwide (Sturm et al. 2005). It is not possible to determine the exact processes at play in our study area, but spatially-structured (Coop & Givnish 2007), micro-climatic (D'Odorico et al. 2010) and biotic (Dullinger et al. 2003) feedback mechanisms seem all plausible.

H4: *Overgrazing*: Overgrazing appears to have accelerated shrub expansion in our study area. However, the temporal and spatial relationships between grazing intensity and shrub expansion suggest that grazing effects were not linear (Fig. 13) and that overgrazing was not the sole driver of shrub encroachment (Fig. 11a).

H5: *Abandonment*: We found no evidence that land abandonment influenced shrub encroachment. However, land abandonment has, as of yet, been a local and recent phenomenon, unlike the widespread rural exodus that resulted in shrub encroachment in the European Alps.

Evidence for a Regime Shift

The observed patterns of shrub encroachment suggest that the alpine zones of Northwest Yunnan are shifting from an herbaceous to a shrub-dominated ecosystem. Regime shifts occur when an ecosystem shifts suddenly from one state to the other, and the shift cannot easily be reversed even when the original environmental conditions are restored (Groffman et al. 2006; Scheffer 2009; Brock & Carpenter 2010). Several lines of evidence suggest that a regime shift occurred. First, multiple causality is the rule in regime shifts, with grassland ecosystems notoriously vulnerable to state changes when both the natural and human systems that affect them change. NW Yunnan’s climate and land use changed substantially since the 1950s (Fig. 14). Typically, prior to a regime shift, perturbations accumulate and start to interact, gradually reducing system resilience before changes arise. After a critical threshold is reached, even a small perturbation can trigger a rapid shift to an alternative stable state. In NW Yunnan, despite increasing stress, we found no record of change in the alpine grasslands from 1950 to the 1980s. However, in the mid- to late 1980s dwarf rhododendron shrubs suddenly established and then spread rapidly.

Initial rhododendron establishment across several alpine territories appears to correspond to two potential triggers. First, May temperatures at 4000 m shifted from sub-zero to above-freezing levels in the mid 1980s, likely reducing snowpack and snowfall. Increases in late-spring soil temperature and moisture can provide a competitive advantage for shrubs (Bjork & Molau 2007; Wipf & Rixen 2010). Second, the 1988 burning ban immediately reduced burning by about 20% (Fig. 13), providing another potential trigger.

Finally, shrubs spread most rapidly in areas with some prior woody vegetation (shrub autocatalysis). A regime shift requires a strong feedback to push a shifting system towards the alternative stable state from which reversion to the original state is difficult. Although burning apparently served historically to effectively control shrubs, burning may no longer do so (Fig. 11b). In fact, dwarf rhododendrons quickly established and colonized many recently burned (<3 years) plots regardless of the original vegetation, supporting the idea that a regime shift occurred.

Broader Impacts

Shifts from one ecosystem to another often start in ecotones, making ecotones important sentinels of change (Hufkens et al. 2009). Our study area straddles the forest-shrub-grassland ecotone at the southern edge of the Himalayas, and may represent a sentinel for the rest of the region.

Vegetation shifts at the scale of the Greater Himalayas would have major implications. The high mountains and plateaus of this region regulate the region's climate while serving as the headwaters of 3 major rivers in Asia. The vegetation shifts we observed may affect regional climate systems and reduce water tables (Euskirchen et al. 2009). This region is also a center of diversity for endemic plants and mammals (Xu et al. 2009). In addition, shrub encroachment degrades meadows, forcing local people to rely more on forest resources (Yi et al. 2007).

Management recommendations

Traditional herding systems sustained both local livelihoods and endemic biodiversity. However, the socio-economic and climatic conditions in NW Yunnan have changed dramatically. Grazing and burning both appeared to be compatible with sustaining alpine ecosystems in the past. However, these activities may be less sustainable now given climate change and heavy shrub encroachment. This leaves it unclear how yak herding is affecting current alpine environments. Although fire was the primary tool for shrub control, it appears to no longer be effective. In addition, while high grazing pressure accelerated shrub expansion, experimental studies show that moderate levels of grazing can mitigate warming-induced shrub expansion (Klein et al. 2007). Herders, officials, and scientists should together perform *in-situ* experimental research to understand grazing thresholds, post-fire regeneration processes, and alternative shrub control strategies. Furthermore, seed collection and seed bank storage, coupled with *ex situ* propagation in botanical gardens throughout the region (Sang et al. 2011), may be needed to preserve threatened endemic plants for future alpine meadow restoration.

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Table 4. Alpine territories visited from 2007-2011. Types of data collected at each site include: G=Ground truth, D=Dendrochronology, FI=Formal Interview, II=Informal Interview

Site	Number of Visits	Timing of Visit	Data Collected
AD	5	Pre-classification	G, D, FI, II
BM	4	Pre-classification	G, D, FI, II
ML	3	Pre-classification	G, FI, II
XD	1	Pre-classification	FI, II
SS	1	Pre-classification	G, FI, II
YR	1	Pre-classification	G, FI, II
HP	1	Pre-classification	G, II
GZ	1	Pre-classification	G, II
YB	1	Pre-classification	G, II
MY	1	Pre-classification	G, II
HS	1	Pre and Post-classification	G, II
QH	2	Pre and Post-classification	G, II
SK	5	Post-classification	G, D, II
AB	1	Post-classification	G, II
TB	1	Post-classification	G, D, II
ZH	1	Post-classification	G, D, II
TC	1	Post-classification	G, II
DX	1	Post-classification	G
LE	1	Post-classification	G
NP	1	Post-classification	G, II
NL	1	Post-classification	G, II

Table 5. Alpine land cover/land use classification accuracy assessment for the Historic (1974-1990) and Recent (1990-2009) classifications.

HISTORIC CLASSIFICATION (1974-1990)

Class	PA	UA	Adj. Area (ha)	±CI (ha)	±CI (%)
Grassland	94%	94%	94566	2245	2%
Shrubland	92%	86%	77296	2962	4%
Forest	99%	95%	257203	3949	2%
Snow	94%	93%	107379	2425	2%
Overall accuracy =			93%		
Khat =			0.93		

RECENT CLASSIFICATION (1990-2009)

Class	PA	UA	Adj. Area (ha)	±CI (ha)	±CI (%)
Encroachment	93%	90%	69999	9624	14%
Grassland	80%	90%	43217	9091	21%
Shrubland	90%	86%	109362	14373	13%
Forest	97%	99%	311401	11290	4%
Overall accuracy =			94%		
Khat =			0.94		

Table 6. Summary of dendrochronology shrub analysis taken from 13 different sites in 5 different alpine territories.

Territory	Site	Land Cover Classification	Dominant Shrub Type	Total % Shrub Cover	Age of All Shrubs Average	SD
Visited Pre-Classification						
BM	B1	Permanent Grass	Rhododendron	50	16	2
	B2	Encroachment	Juniper	60	18	6
	B3	Permanent Shrub	Rhododendron	50	21	4
AD	A1	Permanent Grass	Juniper	6	24	14
	A2	Encroachment	Rhododendron	35	22	8
	A3	Permanent Grass	Juniper	3	18	14
Visited Post-Classification						
SK	S1	Encroachment	Rhododendron	65	4.7	2
	S2	Permanent Shrub	Rhododendron	100	23	8
	S3	Encroachment	Rhododendron	70	17	8
TB	H1	Encroachment	Rhododendron	100	15	7
	H2	Encroachment	Rhododendron	90	20	9
	H3	Encroachment	Rhododendron	90	14	2
ZH	T1	Encroachment	Rhododendron	75	14	6
Average age of all shrubs					**19**	**11**

Figure 9. a) The location of the study area. b) Close-up of land cover/land use change from 1990 to 2009 in one alpine territory. c) Land cover was stable from 1974 to 1990, but grasslands shrunk dramatically from 1990 to 2009 while shrublands expanded.

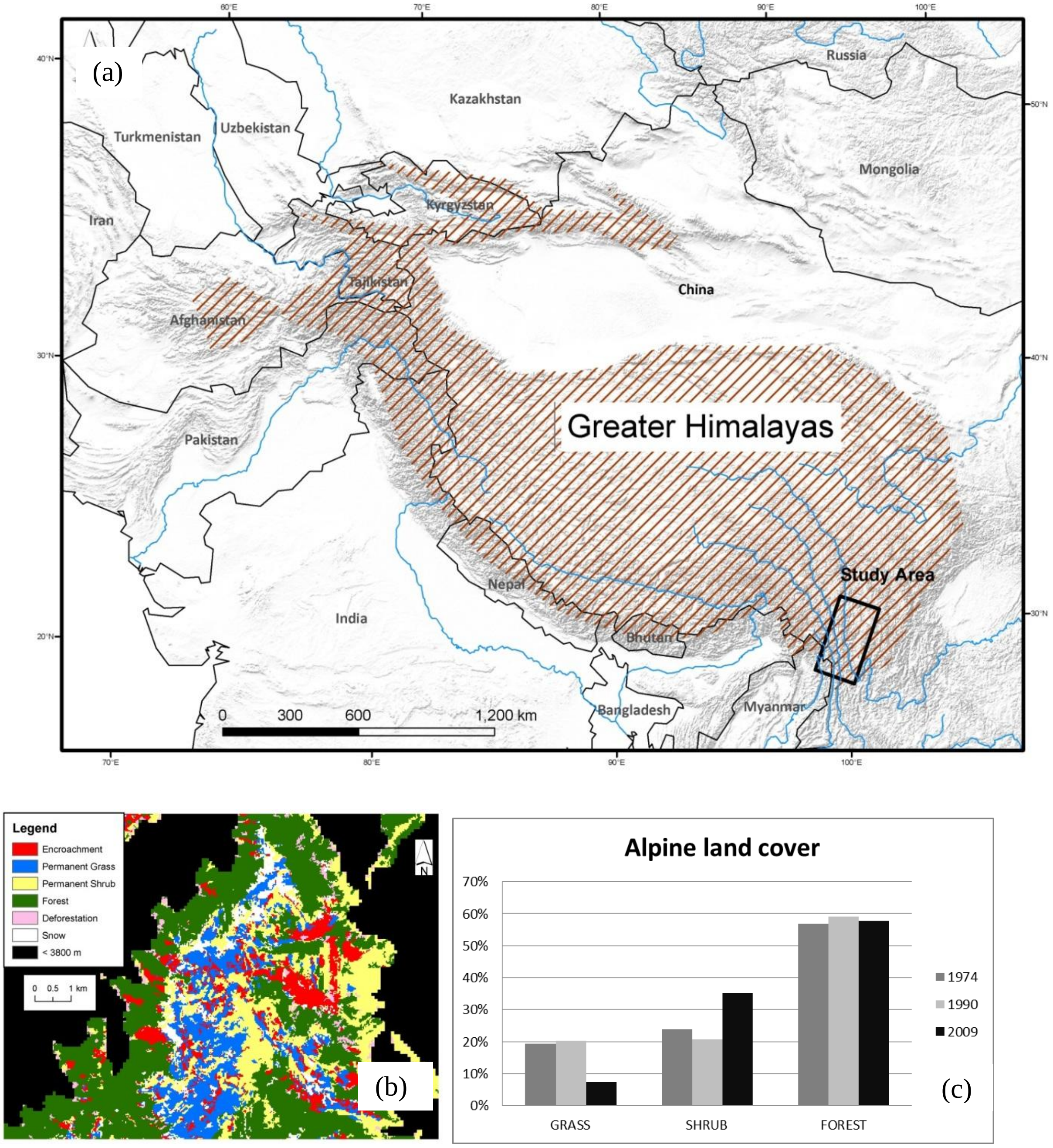


Figure 10. a) Encroachment rates were highly variable throughout the study area, with the northern and eastern regions suffering the highest encroachment rates. b) Shrub encroachment rates (E-Rate) were positively correlated with the proportion of spring snow cover (Snow) and negatively correlated with proportions of woody vegetation in 1990 (Woody). Encroachment was related to c) elevation and d) aspect, with meadows in lower elevations of the alpine zone (4000-4200 m) and drier south facing slopes more subject to encroachment.

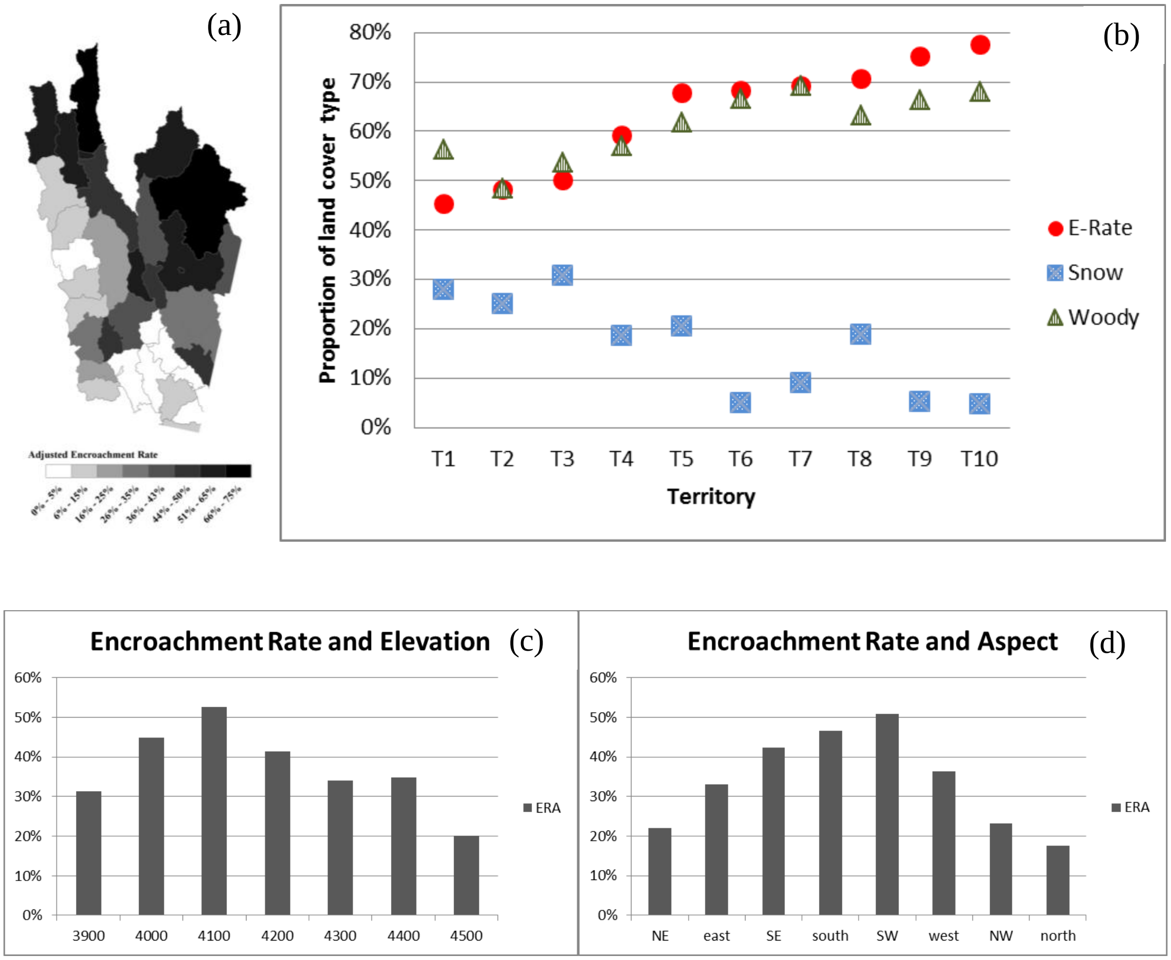
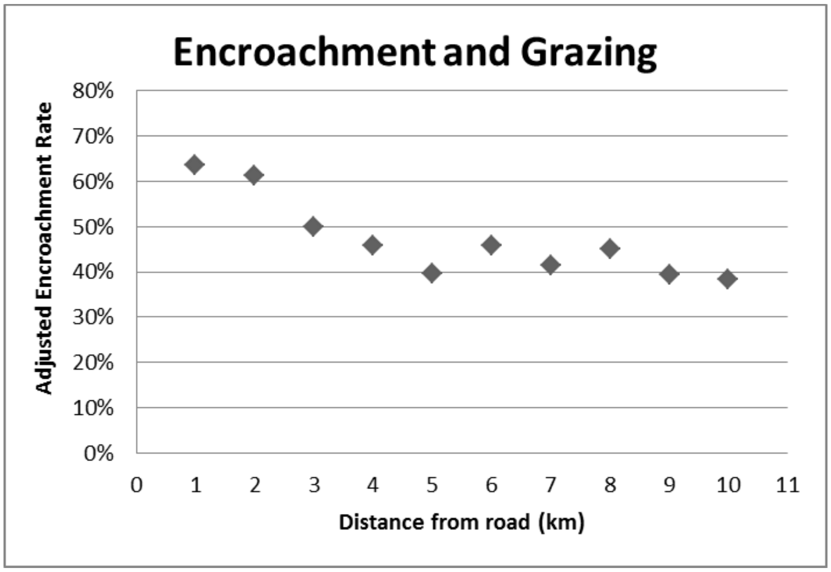


Figure 11. a) Higher grazing intensities close to the national highway increased shrub encroachment rates; b) Relationships between encroachment and burning, and encroachment and abandonment indicate that recent local-scale burning and pasture abandonment did not influence regional patterns of shrub encroachment.

(a)



(b)

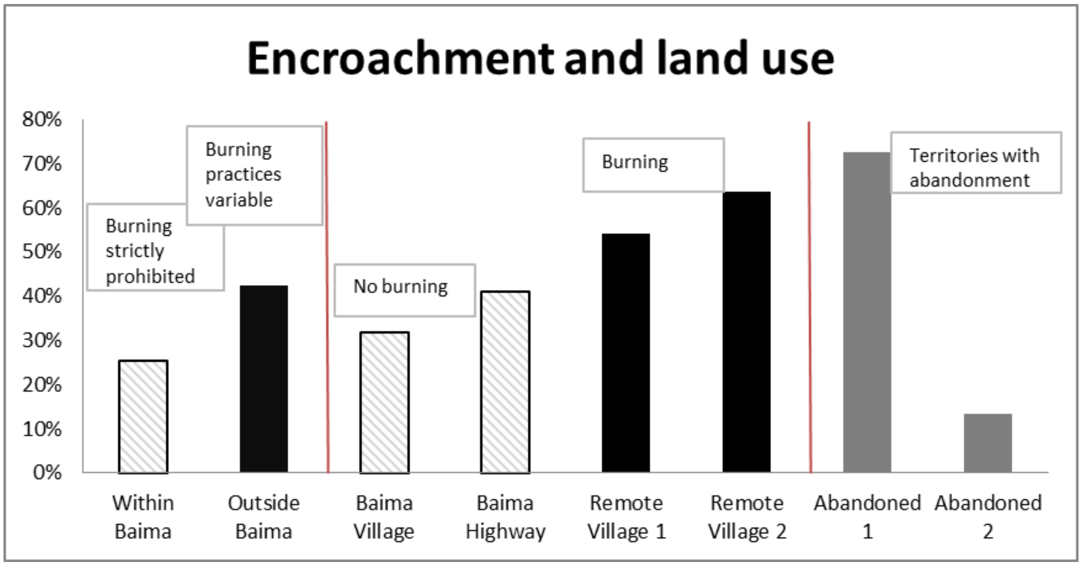


Figure 12. Total a) snowfall (mm) and b) rainfall (mm) measured at Deqin climate station (3320m). Estimated average c) daily low and d) daily high temperatures (°C), at 3800m, 4000m, 4200m and 4400m.

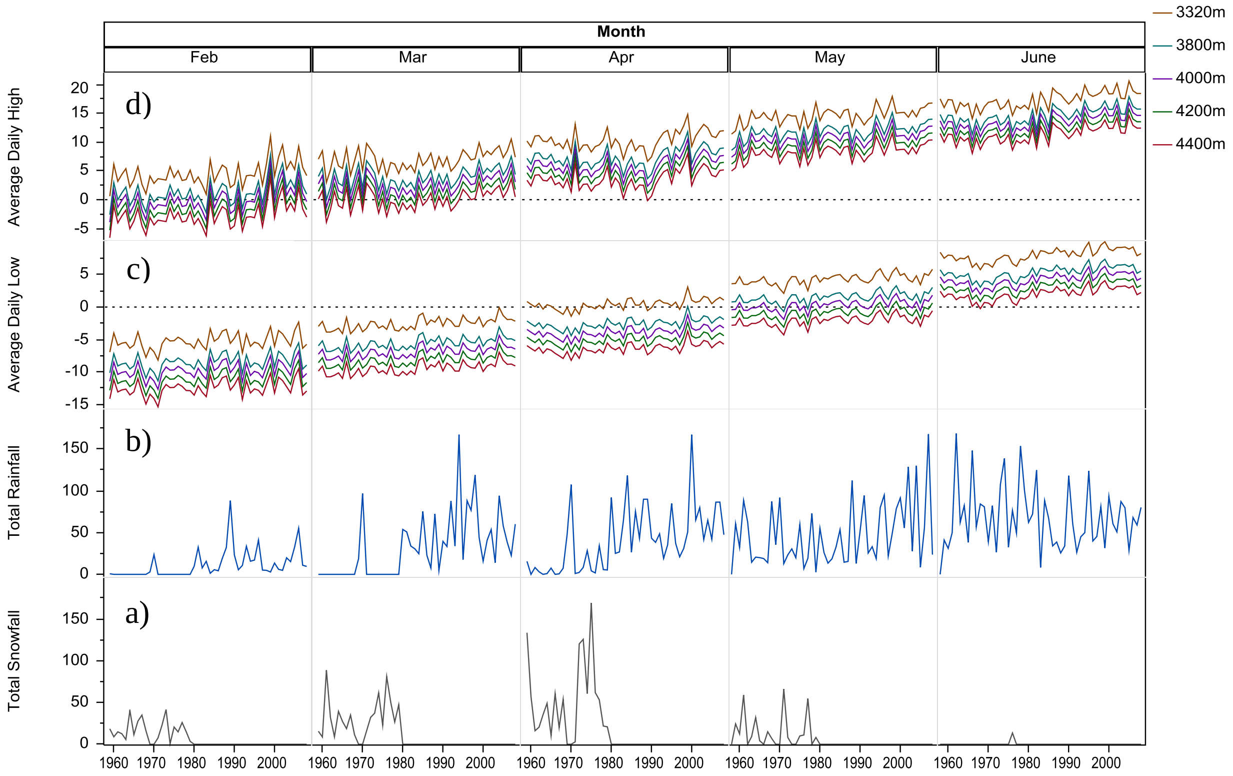


Figure 13. A reconstruction of land cover, land use and climate change from 1951 to 2009.

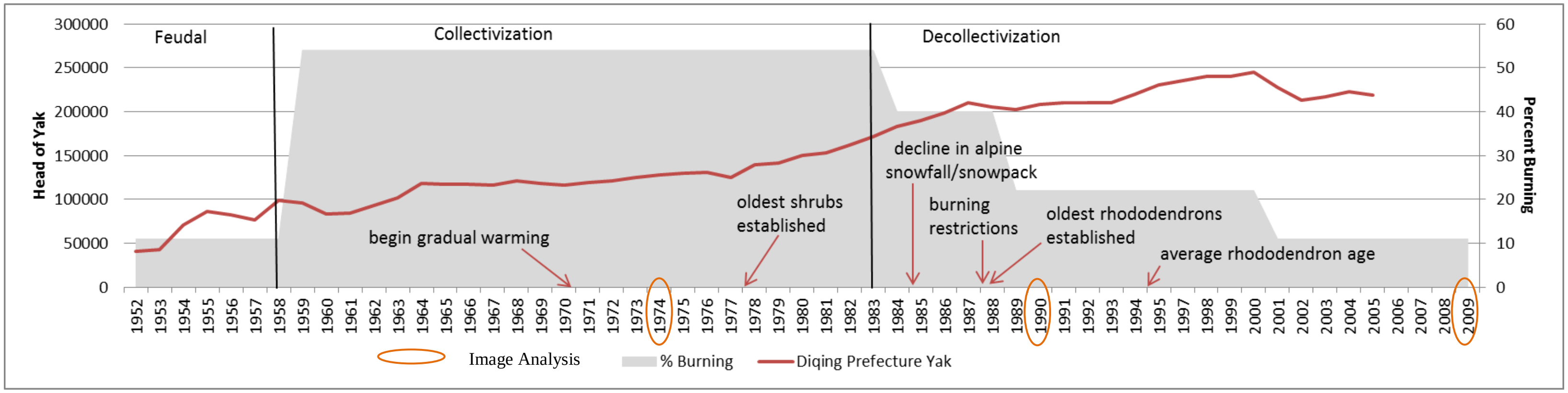
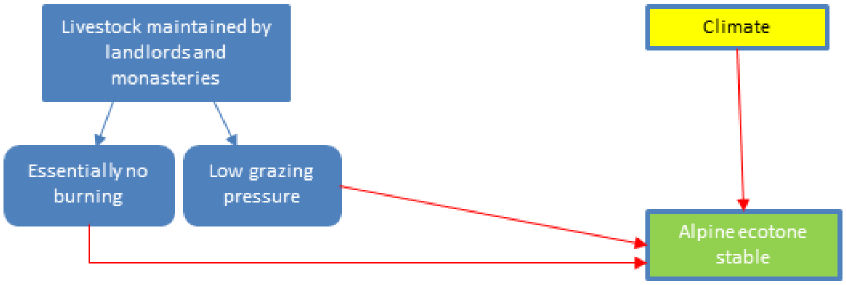
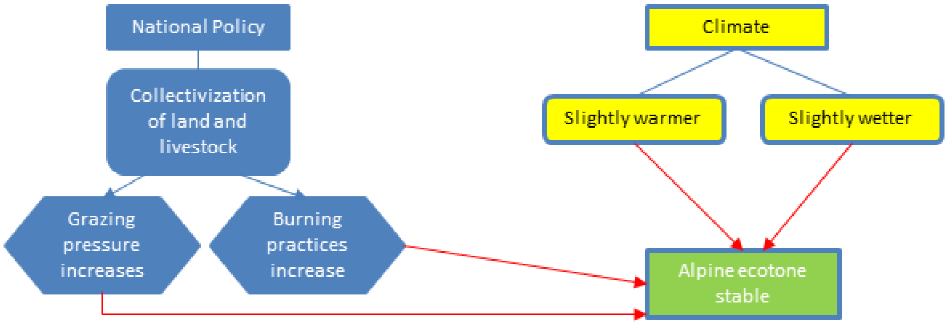


Figure 14. Interactions between social, climate and alpine sub-systems during a) the feudal era, b) the collectivization period and c) following decollectivization.

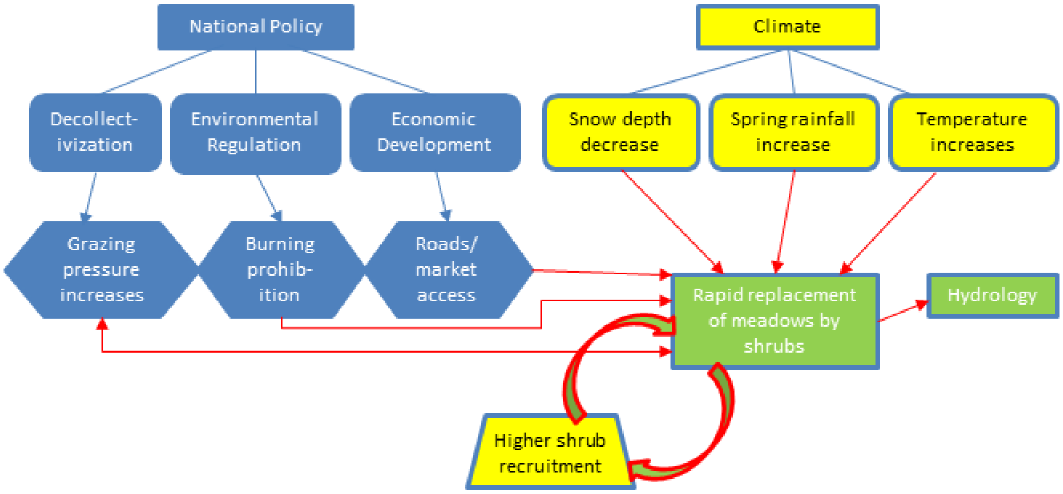
(a) Feudal Era (pre-1958)



(b) Collectivization (1958-1983)



(c) Decollectivization (1983-present)



Chapter 3: Sacred forests are keystone structures for forest bird conservation in southwest China's Himalayan mountains

Abstract

Identifying and protecting "keystone structures" is essential to maintain biodiversity in an increasingly human-dominated world. Sacred areas, i.e. natural areas protected by local people for cultural or religious regions, may be keystone structures for forest birds in the Greater Himalayas, but there is limited understanding of their use by bird communities at different spatial scales. We surveyed birds and their habitat in and adjacent to six Tibetan sacred forest patches in northwest Yunnan China, a biodiversity hotspot. Our goal was to understand the ecological and conservation role of these remnant forest patches for forest birds. We found that sacred forests supported a significantly different bird community than the surrounding matrix, and had higher bird species richness at plot, patch, and landscape scales. Furthermore, while we encountered a single matrix bird community, the sacred forest patches exhibited high within- and between-patch heterogeneity, and supported several distinct sacred forest bird communities. Finally, we found an increased use of the sacred forests and their edges during 2010, a severe drought year in Yunnan Province, indicating that sacred forests may serve as refuges during extreme weather years. While bird community composition was primarily driven by the vertical structure of the vegetation, plots with the largest diameter trees and the most bamboo coverage had the highest bird diversity and abundance, indicating the importance of protecting old-growth forest ecosystems for Himalayan forest birds. Old-growth forest clearing and degradation have accelerated throughout the Himalayas, and our results offer hope for forest birds, in that existing sacred areas protect a variety of habitat niches and increase bird diversity at multiple spatial scales. As population growth and rapid economic development continues throughout the

Himalayas, sacred areas are an important opportunity for biological conservation at the local, landscape and regional scales.

Introduction

As humans consume an ever-increasing proportion of the Earth's resources, biodiversity declines at an accelerating rate (Chapin et al. 2000; Myers & Kent 2003; Dobson et al. 2006). Identifying and protecting "keystone structures" may be essential to preserve biodiversity in an increasingly human-dominated world. Keystone structures are discrete spatial features that maintain biodiversity, at multiple spatial scales, despite being small in proportion to the entire ecosystem (Tews et al. 2004). For example, forest gaps, large trees, and temporary wetlands are keystone structures whose presence adds heterogeneity to the resources available in the landscape, facilitating greater species richness (Belsky & Canham 1994; Tews et al. 2004; Manning et al. 2006; Stagoll et al. 2012).

Sacred forests, i.e., natural areas protected by local people for cultural or religious reasons, are important for biodiversity conservation (Bhagwat & Rutte 2006; Dudley et al. 2009). Sacred forests are numerous and dispersed throughout the landscape, located in a broad range of topographic and micro-climatic conditions, and range in size from a single hectare to thousands of square kilometers (Ormsby 2011). As such, they may serve multiple ecological functions, including as corridors, refugia, and source habitats (Bhagwat & Rutte 2006). Sacred forests are often embedded within human-dominated landscapes, managed by communities, and highly resilient to social change (Dudley et al. 2009). They may be critical components of protected area networks, especially in densely populated landscapes. Sacred forests may be keystone structures for biodiversity in traditional landscapes throughout the world, but we have little understanding of the their spatial extent and configuration, especially in less-studied biodiversity hotspots.

The traditional land management systems that sustain sacred forests may create optimum conditions for species diversity at multiple spatial scales. Sacred forests are not necessarily pristine, and often experience a within-forest gradient of human disturbance (UNESCO-MAB 2003), where a variety of organisms can respond to and thrive in variable resource conditions (Belsky & Canham 1994). In addition, sacred forests are typically embedded in landscapes with matrix habitats that are hospitable to at least some species, and thus conventional assumptions of patch size and fragmentation effects (MacArthur & Wilson 1967) may not apply (Ricketts 2001; Bhagwat et al. 2005; Prugh et al. 2008; Watling et al. 2011). Furthermore, the edges between sacred forests and their matrix are often not abrupt barriers, but a gradient of disturbance to levels characteristic of the surrounding matrix. These edges may serve as “ecotones”, facilitating ecological interactions between the patch and the matrix, and offering supplementary resources not available in the core habitats (Holland et al. 1991; Ries et al. 2004; Hufkens et al. 2009). However, despite their importance for species diversity and distribution patterns, we have little understanding of how sacred forests are influenced by patch size, edge effects, and interactions between the patch and matrix habitats.

The Greater Himalayas contain three biodiversity hotspots (Myers et al. 2000; Barbhuiya et al. 2010) and forest birds are of special conservation concern (Renner 2011). The region exhibits high levels of bird diversity and endemism (Acharya et al. 2011; Renner & Rappole 2011), and rank highest in global assessments of threatened bird species richness (Grenyer et al. 2006). Forest degradation has accelerated in recent decades (Spehn et al. 2010; Brandt et al. 2012), destroying bird habitats (Dumbacher et al. 2011). Despite a serious threat to regional bird diversity, we have little understanding of the consequences of recent landscape change to Himalayan forest bird communities. (Renner 2011).

Many forest bird species in the Greater Himalayas follow a sino-Himalayan distribution (Fig. 1a), which includes the Himalayan range, the mountains of southwest China, and the

Qinghai Tibetan plateau (Renner 2011; Renner & Rappole 2011). This same area is home to several ethnic minority groups that recognize sacred areas as part of their religion, including sacred beyuls, which protect entire valleys (Mallarach 2008), sacred mountains that range from 10s to 100s of km^2 (Peng et al. 2003), and village-level sacred forests and groves that vary in size from 1 to 1000 ha (Peng et al. 2003; Barbhuiya et al. 2010). These sacred areas may be critical for conservation throughout this rapidly changing region (UNESCO-MAB 2003; Xu et al. 2005; Salick et al. 2007; Barbhuiya et al. 2010). However, their ecological importance as well as how bird diversity varies among sacred and non-sacred habitats across multiple spatial extents is not well understood.

Our overarching objective was to understand the ecological role of sacred forests for the conservation of Himalayan forest birds. We studied bird communities within and outside of Tibetan sacred forests in northwest Yunnan, China, with the following specific objectives:

1. Determine if bird community composition and diversity is different within sacred forests compared to the surrounding matrix,
2. Identify the critical habitat characteristics structuring bird community, diversity and abundance, and
3. Investigate how patch size and edge habitats influence bird community composition, diversity and abundance patterns.

Study Area

Our study area is in Shangrila, northwest (NW) Yunnan Province China (Fig. 1a). NW Yunnan is in the southeastern sub-Himalayan mountains, bordering Myanmar, Tibet and Sichuan Province. Three major rivers (the Yangtze, Mekong, and Salween) create steep gorges, with elevations ranging from 1,800 to 6,740 m, creating a large array of ecological niches in a relatively small area. NW Yunnan is a corridor between the tropics of Southeast Asia, the Himalayan mountains, and the Qinghai-Tibet Plateau, and is located within one of the Himalayan region's biodiversity hotspot, the Mountains of Southwest China (Myers et al. 2000).

NW Yunnan has great importance for local, regional, and global bird diversity. First, it is a center of bird endemism (Lei et al. 2003). Second, it is along a major thru-way of the East Asian migratory flyway and provides important stop-over habitat for long-distance migrants (Fig. 1a) (Cheng 1987; McClure 1998; Wang et al. 2000). Third, it lies at the confluence of the Palearctic, Himalayan, and Indo-malay zoogeographic regions, and thus provides habitat for birds with distinct ecological and evolutionary histories (Renner 2011).

NW Yunnan's avifauna is one the poorest understood on Earth (Renner & Rappole 2011), and it is likely that forest birds face serious threats. Large expanses of NW Yunnan's forests were clear-cut by state logging companies from the 1960s through the 1990s to fuel China's national development (Harkness 1998; Morell 2008). Despite a ban on commercial logging in 1998 and heavy investment in reforestation and protected areas, (Liu et al. 2008), old-growth forests continue to be logged and the ecological integrity of the new forests is unclear (Xu 2011; Brandt et al. 2012).

NW Yunnan is home to several ethnic minority groups that recognize sacred areas as part of their religion (Xu et al. 2005). In particular, Tibetans have a complex sacred site system that includes Tibetan sacred mountains, which are few in number but large in size (tens to hundreds of square kilometers) but also hundreds of community-based sacred forests that are smaller (1 to 1,000 ha) and at lower elevations (Peng et al. 2003). Vegetation communities in Tibetan sacred forest sites have higher vegetation species richness, diversity, and endemism than randomly selected non-sacred sites (Anderson et al. 2005), and larger and denser trees (Salick et al. 2007). However, very little is known about the importance of sacred forests for non-plant taxa.

Methods

Avifaunal sampling

Breeding bird point count surveys were conducted at six sacred forest sites that ranged in size from 13 to 75 ha (Fig. 1b). We surveyed a total of 62 plots, including 35 within the sacred forests, 9 edge plots (60-m away from the boundary of the sacred forest), and 18 matrix plots (260-m and 520-m from the boundary of the sacred forest patch) (Fig. 1c). Plots were placed at least 200-m apart, along walking paths when possible.

At the plot center, we surveyed birds at least 2 times per year during the breeding season (May 18-June 30) in 2010 and 2011. Surveys consisted of 50-m radius, 10-minute point counts following standardized methods (Ralph et al. 1993), and included an estimate of distance to bird using a hand-held laser rangefinder (Buckland et al. 2001). All point counts were conducted by JSB and EMW.

Habitat structure

Vegetation composition and structure was measured at each point-count station according to standardized protocols (Ralph et al. 1993; Martin et al. 1997) in June to August of 2010. Within each point-count station, measurements were made in four 5-m radius subplots (Ralph et al. 1993). The initial sub-plot was located at the center of the point-count station, and the remaining three subplots were located 30-m away from the center subplot at 0º, 120ºand 240º. Slope and aspect were measured at the center of each subplot. Canopy coverage of live trees and snags was measured using a densitometer at the four corners (north, east, south, and west) of the

5-m subplot. Basal area was measured at the center point of each subplot using a factor 10 or factor 20 square feet-per-acre prism. Foliage height diversity in 0.3-m sub-sections was measured at the four corners of each of the four 5-m subplots. Ocular estimates for the sub-canopy layer (0.5-5-m) and the ground layer (0-0.5-m) were collected at each subplot. Also, the entire 50-m radius plot was sampled and all species of trees and shrubs encountered were identified to species, and the heights of the three tallest trees and diameter-at-breast height for the three largest-diameter trees were recorded.

Data Analysis

Differences between sacred forest and matrix habitats

We estimated differences in detection probabilities among habitats, and between years by performing density adjustments for those species for which we had sufficient observations (>75) in each year (Buckland et al. 2001). For this analysis, we included bird observations from the edge plots with those from the matrix plots in order to estimate a species detection curve representing either sacred or non-sacred (i.e., edge and matrix plots) focal bird density. We chose to lump edge plots with matrix plots because the habitat characteristics were more similar between edge and matrix than edge and sacred forests and we felt that these habitat similarities resulted in similar bird detectability. There was a single species with sufficient observations in both 2010 and 2011 (Blyth’s Leaf Warbler), and an additional species with sufficient observations in 2010 only (Hume’s Leaf Warbler). Three models were used to fit the detection function (the probability of sighting with perpendicular distance from the transect line): uniform, half-normal, and hazard-rate (Buckland et al. 2001). The best model was selected using the Akaike's Information Criterion (AIC) (Burnham & Anderson 2002).

We compared bird diversity and abundance among habitat types and between years at three different spatial scales: plot, patch and landscape. At the plot scale, we compared raw values of observed bird species richness, Shannon index, and absolute abundance in sacred, matrix and edge habitats. Differences among habitat types were tested using a one-way analysis-of-variance (ANOVA) and pairwise comparisons (Tukey multiple comparisons of means). For the patch and landscape scale analyses, we aggregated edge and matrix plots into a single "matrix" class, resulting in sufficient data to build rarefaction curves and account for differences in sampling effort and individuals observed. At the patch scale (n=6), we compared sacred and matrix habitats using observed species richness, rarefied species richness and rarefied Shannon index, standardized by the number of individuals, derived from sample-based rarefaction curves constructed in EstimateS v.7 (Colwell 2009). Differences at the patch scale were tested using paired two-sample student's *t*-tests. At the landscape scale, we calculated rarefied species richness, rarefied Shannon index and estimated total species richness (using the Jackknife 1 estimator) in EstimateS (Colwell 2009; González-Oreja et al. 2010). Patterns of bird diversity according to the Shannon Index were consistently similar to raw and rarefied species richness, and therefore only raw and rarefied species richness were reported here.

To determine whether bird species assemblages differed among habitats and sacred forests, we removed rare species (less than 3 observations per year) from our multivariate dataset and computed a Bray-Curtis similarity index on both square root-transformed abundance data and on presence/absence data (Carr 1997). Resulting patterns were similar, and here we reported the results from the square root-transformed abundance data. Using the Bray-Curtis similarity matrix, we used non-metric multidimensional scaling (NMDS) to create a graphical representation of the resemblance matrix. NMDS plots individual sites on a two-dimensional ordination plot with similar sites appearing closer together and dissimilar sites plotted further apart (Kruskal 1964). Bird assemblage differences among a) habitats and b) sacred forests were tested using an analysis of similarity (ANOSIM). We used a Bonferonni adjusted alpha value for pairwise comparisons among the three habitats ($p = 0.05/3 = 0.016$). NMDS and ANOSIM analyses were performed using PRIMER v6 (Clarke & Gorley 2006).

Habitat selection

We performed two different analyses to understand bird habitat selection in our study area. First, we used linear regression to identify habitat factors that contributed to high bird diversity and abundance at the plot scale. We performed simple linear regression to test the significance of individual factors for raw species richness, Shannon diversity, and abundance values at the plot scale in both 2010 and 2011. Following this analysis, we chose a candidate set of *a-priori* uncorrelated environmental variables thought to be most influential in determining bird community structure, diversity, and abundance (Appendix 1), and performed a step-wise multiple linear regression models to determine the most important environmental variables influencing bird diversity and abundance. The best model was selected using the Akaike's Information Criterion (AIC) (Burnham & Anderson 2002). To investigate the importance of patch size on species richness, we performed Pearson’s correlation analysis between patch size and raw species richness. These analyses were performed in the statistical package R version 2.8.1.

To identify the most prominent habitat gradients and the most important variables driving those gradients, we performed principal components analysis (PCA). PCA is an unconstrained ordination technique suited for multivariate habitat data. It uses linear modeling to reduce multiple variables (“components”) to a small number of the most important (or “principal”) habitat gradients. PCA provides information on the strength of the reduced components to explain the variability in the data, and the contribution of individual habitat variables to the principal components. Prior to PCA analysis, we normalized the environmental variables and inspected cross-correlation matrices to identify and remove variables with high cross-correlations. We eliminated all but seven variables from the environmental dataset, choosing variables that were both uncorrelated and known to be important for bird communities from the regression analysis. PCA was performed using PRIMER v6 (Clarke & Gorley 2006).

To understand the relationship between the prominent habitat gradients and bird community composition we performed canonical correspondence analysis (CCA). Canonical correspondence analysis is a constrained ordination technique, which constrains habitat gradients by the bird species data. As with the PCA, we eliminated all but seven variables from the environmental dataset, choosing variables that were both uncorrelated and known to be important variables determining bird diversity and abundance. CCA was performed in the software PC-ORD 6.04.

Results

Differences between sacred forest and matrix habitats

Habitat

We surveyed 62 plots that captured a wide range of variability in vegetation disturbance, age, structure and vegetation species composition (Appendix 2, Fig. 1b). PCA identified two prominent habitat gradients in our study area (Fig. 2a). The first axis (eigenvalue=3.59, explaining 44.8% of the variance) corresponded to differences between the sacred and matrix habitats, which had significantly different vegetation composition and structure in the canopy, sub-canopy and ground layers (Appendix 1). The first axis had high levels of vertical structure (foliage height diversity), canopy cover and downed logs on the left and high proportions of bare soil and multiple-stemmed woody vegetation in the ground layer (0 – 0.5 m) on the right. The second axis of the PCA (eigenvalue=1.51, explaining 18.9% of the variance) represented a within-sacred forest gradient. Sacred forests with dense understory vegetation (i.e. high proportions of sub-canopy coverage and bamboo) corresponded to high values of Axis 2, while those with a more open understory (high herbaceous cover and downed logs in the ground layer) corresponded to low values (Fig. 2a).

Birds

We observed a total of 81 species exhibiting breeding behavior (Appendix 3). Detection probability was 5% to 21% higher in the matrix than in the sacred forests, indicating that the raw data underestimates species richness and abundance more in the sacred forests than in the matrix habitats. Detection probability was 10% to 17% lower in 2010 than in 2011, and therefore we performed all analyses separately for the two years. We used raw counts for the statistical analyses because sample size was too low to estimate density for most species.

Species composition of the bird communities overlapped between the sacred, edge, and matrix habitats (Fig. 2b), but still exhibited statistically significant patterns (NMDS, 2D stress=0.18, ANOSIM Global $R = 0.254$, p=0.001). Edge and matrix plots shared the same bird community (p=0.51), while sacred forest bird communities were distinct from edge (R=0.227, p=0.005) and matrix (ANOSIM, $R = 0.37$, p=0.001) communities. Furthermore, different sacred forests supported distinct bird assemblages. Species composition was significantly different in five of 15 pairwise comparisons (ANOSIM, Global R=0.22, p=0.001), indicating that the six sacred forests provided habitat for more than one bird community (Fig. 2b).

We measured differences in bird species richness at three spatial scales. At the plot scale, mean species richness was higher in sacred forests (13.3 ± 3.1) than in edge (11.1 ± 2.9, $p = 0.084$) and matrix (9.64 ± 1.9, $p < 0.001$) (Fig. 3a). Bird abundance was slightly higher in sacred forests than in edge or matrix, but these differences were not significant ($p = 0.53$ and 0.33, respectively) (Fig. 3b).

Since edge and matrix were always similar in terms of bird species composition and diversity, we aggregated edge and matrix plots into a single “matrix” class at the patch and landscape scales, which allowed us to build rarefaction curves and thereby adjust for differences in sampling effort and observed individuals. At the patch scale, sacred forest patches had higher mean rarefied species richness (28 ± 6.8) than the matrix (21 ± 5.8) ($p < 0.001$) (Fig. 3c). At the landscape scale, bird species accumulated fastest in the sacred forest habitat (Fig. 3d). Rarefied species richness (rarefied to 824 individuals) was higher in the sacred (59 ± 4.1) than the matrix habitat (53 ± 4.1).

Habitat Selection

Habitat variables associated with high bird diversity and abundance

Four variables were selected during stepwise regression as important predictors of species richness: the diameter of the largest tree, percent bamboo, percent ground cover of downed logs, and percent sapling cover (Stepwise Regression AIC = 112, Adj. $R^2 = 0.46$, $p < 0.0001$). In simple linear regression models to predict bird abundance, only one variable, the diameter of the largest tree, was a significant predictor of abundance (Adj. $R^2 = 0.20$, $p = 0.00018$).

Habitat variables structuring breeding bird communities

The canonical correlation analysis indicated that habitat gradients and species composition were highly correlated (Axis 1 Pearson's $r > 0.95$, Axis 2 Pearson's $r = 0.718$, Fig. 4). Axis 1 (Eigenvalue = 0.480, explaining 14 % of the variance in the species data) captured a gradient between birds associated with the matrix habitats at low values, and those associated with the sacred forest habitats at high values. Foliage height diversity had the highest correlation with Axis 1 ($r = -0.525$), indicating that it explained the highest variance in the species data. Axis 2 (Eigenvalue = 0.114, explaining 3.3% of the variation in the species data) captured a gradient of within-sacred forest habitats. Birds associated with dense understory cover had high values of Axis 2, while birds associated with high canopy cover and downed logs had low values.

Edge effects and patch-matrix interactions

Our plot-scale analyses indicated that edge was similar to matrix in terms of bird community, diversity, and abundance. To investigate whether edges influenced bird diversity patterns we regressed plot-scale bird richness and abundance with plot distance from the edge of a sacred forest. Within sacred forest patches, species richness increased with distance to the edge of the sacred forest patch ($R^2 = 0.11$, $p=0.06$). For the matrix, the opposite relationship was found. Matrix bird diversity was highest at the edge, but decreased with distance from the sacred forest ($R^2=0.17$, $p=0.03$). There was no relationship between bird abundance and distance from the edge.

We analyzed the correlation between patch size and bird species richness for both the sacred and matrix plots of each sacred forest. There was a positive and significant correlation between patch size and raw species richness in the sacred forest patches ($r=0.79$, $p=0.03$) and in the matrix plots associated with each sacred forest patch ($r=0.80$, $p=0.05$). However, when standardizing for sample effort and number of individuals, the correlations between patch size and rarefied species richness were not significant.

Differences between 2010 versus 2011

Plot-scale abundance was significantly higher in all habitats (sacred, matrix, and edge) in 2010 compared with 2011 (Fig. 5a). Plot-scale bird richness was significantly higher in 2010 than in 2011 in the sacred forests and edge plots, but not in matrix plots (Fig. 5b). Total (estimated) species richness curves showed that total species richness was higher in 2010 compared to 2011 for all habitats (Fig. 5c). However, when accounting for sampling effort and individuals observed, species accumulated at the same rate in both years in both the matrix and sacred habitats, but species accumulated at a much faster rate in edge habitats in 2010 versus 2011.

Discussion

Differences between sacred forest and matrix habitats

Our results indicated that Tibetan sacred forest conserved old-growth forest characteristics and protected unique forest bird communities in the Chinese Himalayan mountains. Bird species composition in sacred forests was distinct from that in the surrounding matrix. In addition, sacred forests had the highest bird abundance and diversity of all habitats at multiple spatial scales and in both years. Since detection probabilities were considerably lower in the sacred forests compared to their matrix, it is likely that the differences in both diversity and abundance were underestimated.

The observed differences in bird communities occurred because sacred forests have been protected from logging, are composed of a mixture of successional and old-growth forests, and are structurally complex. In contrast, the surrounding matrix has been subject to a range of high-intensity subsistence land-use pressures including logging, non-timber forest resource collection, and grazing, that have resulted in a relatively homogeneous matrix habitat.

As such, sacred forests exhibit several characteristics of keystone structures (Tews et al. 2004). First, sacred forests are highly heterogeneous patches embedded in a landscape with a comparatively homogeneous vegetation structure. Second, sacred forests maintained higher bird and vegetation diversity despite being small in proportion to the adjacent matrix habitats that make up the majority of the Shangrila landscape. Third, the maintenance of diversity was not dependent on a particular tree species or forest community, but instead, bird diversity was associated with structural characteristics that increased habitat heterogeneity. Finally, the positive relationship between habitat heterogeneity and bird diversity was consistent across multiple spatial scales.

Sacred forests experienced a gradient of human disturbance that corresponds with distance from the sacred area's focal point (e.g., temple, summit, altar) as well as topography. As a result, single sacred forest patches contained multiple habitat types, while matrix plots were relatively homogeneous. Within-patch heterogeneity in keystone structures is important to create an environment where a variety of organisms can respond to - and thrive in - the variable resource conditions of the patch (Belsky & Canham 1994). The structural and compositional habitat heterogeneity within sacred forests presumably increases resource abundance and availability which are likely factors for why bird diversity was greater in sacred forests than matrix habitats. At the landscape scale, bird diversity was higher in sacred forests because of between-forest heterogeneity (McGarigal & McComb 1995). The sacred forest patches were relatively close to each other (<20 km) and at similar elevation (3200 to 3800 m), but contained a variety of habitat types. As a result, the sacred forests supported a wide range of bird species, and differed in their community composition.

Habitat Selection

Sacred forest plots had higher vertical complexity, and retained large trees, bamboo groves and downed logs not present in the matrix. The relatively large range in vertical complexity (i.e., foliage height diversity, MacArthur & MacArthur 1961) of our survey plots was the main characteristic structuring bird species composition, while old-growth trees, bamboo groves and downed logs were the most important factors determining high species richness and abundance.

Our results indicate the importance of mature forest ecosystems over secondary or disturbed forest habitats. In some ecosystems, bird diversity peaks in secondary growth stages (Keller et al. 2003; Schieck & Song 2006), while in other ecosystems, primary forests support unique, high-diversity bird communities (Ries et al. 2004; Barlow et al. 2007; Ding et al. 2008). Old-growth forests, including large trees and bamboo groves (a native understory component), have been heavily exploited for both subsistence and commercial use in this region (Morell 2008; Xu 2011; Brandt et al. in press). Our results suggest that the protection of Tibetan sacred forests, and other forests that retain old-growth characteristics, should be a high priority for forest bird conservation in this biodiversity hotspot.

Bird diversity at the plot scale was also positively associated with high proportions of sapling cover (Table 2). Post-logging tree regeneration in the matrix is limited due to logging practices that leave no mature trees as sources for natural regeneration, and by grazing which destroys any seedlings and saplings which do grow, allowing shrub and scrub forest communities to persist for decades following forest clearing (Xu & Melick 2007; Urgenson et al. 2010). Land management practices that encourage tree-sapling regeneration in the matrix would benefit forest bird communities.

Influence of patch size and edge habitats on bird community patterns

While even the smallest remnant forest patch that we surveyed (13 h) had a bird community that was distinct from - and more diverse than - the surrounding matrix, our results suggest that large patches are especially important for forest bird communities. First, larger patches supported more bird species, likely because they contained more habitat niches (MacArthur & Wilson 1967) and supported more individual birds, than small patches (More Individuals Hypothesis; (Srivastava & Lawton 1998)). Furthermore, species richness increased with distance from the edge into the interior of the sacred forests.

Second, our results indicated that sacred forests contained higher quality habitat compared to the edges, and, generally speaking, large patches have a higher ratio of core forest to edge habitats compared to small patches (Turner et al. 2001). In some ecosystems, edges serve as an ecotone, providing supplementary resources that are not present in the core or matrix habitats, which results relatively high species richness and abundance and distinct species assemblages at edges (Holland et al. 1991; Ries et al. 2004; Hufkens et al. 2009). However, our bird community analysis indicated that sacred forest edges were not a distinct habitat, but rather an intermediate and overlapping zone between the sacred forest and matrix habitats. In addition, although edge plots had slightly higher richness and abundance compared to the matrix plots, these differences were not statistically significant,

Differences between 2010 versus 2011

We found clear inter-annual differences in bird diversity and abundance at our sites. For example, bird abundance was higher in all habitats in 2010, but only in sacred and edge plots did we encounter higher bird diversity in 2010. Plots in the interior of sacred forests had especially high abundance and diversity in 2010, but when accounting for the differences in sampling effort and abundance, only the edge plots had a higher rate of species accumulation. Yunnan province received 60% less rainfall than normal during the 6 months leading up to the 2010 breeding season (Qiu 2010; Lu et al. 2012). The typical effect of a drought is to reduce bird abundance and diversity (Albright et al. 2010). We found the opposite pattern, with an increased use of the sacred forest patches and edge habitats during 2010, indicating that sacred forests may serve as refuges during extreme weather years.

Management Implications

Sacred forests are apparently keystone structures for forest birds in NW Yunnan, and, potentially, throughout the Himalayas. Faced with rapid land use and climate change, it is important to establish reserve networks that cover the entire elevation and aspect continuum present in this highly diverse mountainous region (Wu et al. 2010). However, the establishment of large, contiguous protected areas is challenging in this densely inhabited landscape. Since the 1980s, China’s protected area system has expanded greatly, adding 1500 nature reserves nationwide, but there is wide acknowledgement that their effectiveness is limited (Li & Han 2000; Liu et al. 2003; Tang et al. 2006). Sacred forests throughout the Greater Himalayan region provide an existing network of native vegetation patches that protect a variety of habitat niches, maintain biodiversity at multiple spatial scales, and may even function as refugia during extreme weather years.

However, neither the ecological role nor the sociocultural mechanisms underlying sacred forests are well understood. Unlike formally-designated protected areas, sacred forests have proven to be remarkably resilient to social change (Dudley et al. 2009). However, cultures evolve, and the current social trend is towards cultural assimilation and the degradation of sacred forests (Barbhuiya et al. 2010; Rutte 2011). Incorporating sacred forests into official protected area networks is not necessarily desirable from the community perspective, as it leads to a loss in legal ownership and property rights (Bhagwat & Rutte 2006). Furthermore, as ecotourism expands, many communities exploit sacred forests for commercial use as touristic destinations (Barbhuiya et al. 2010), which may protect the forests from extractive use, but can lead to a fundamental change in their meaning, management and ecological function. Efforts to understand the underlying cultural values and mechanisms of sacred areas are necessary if they are to continue to play their keystone role of supporting biodiversity in this rapidly changing region.

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Figure 15. a) Location of Shangrila within the Greater Himalayan region. b) We sampled six sacred forest patches. c) Sampling plots were placed within and outside of sacred forests, with matrix plots placed approximately 60m, 260m, and 510m away from the edge of the forests.

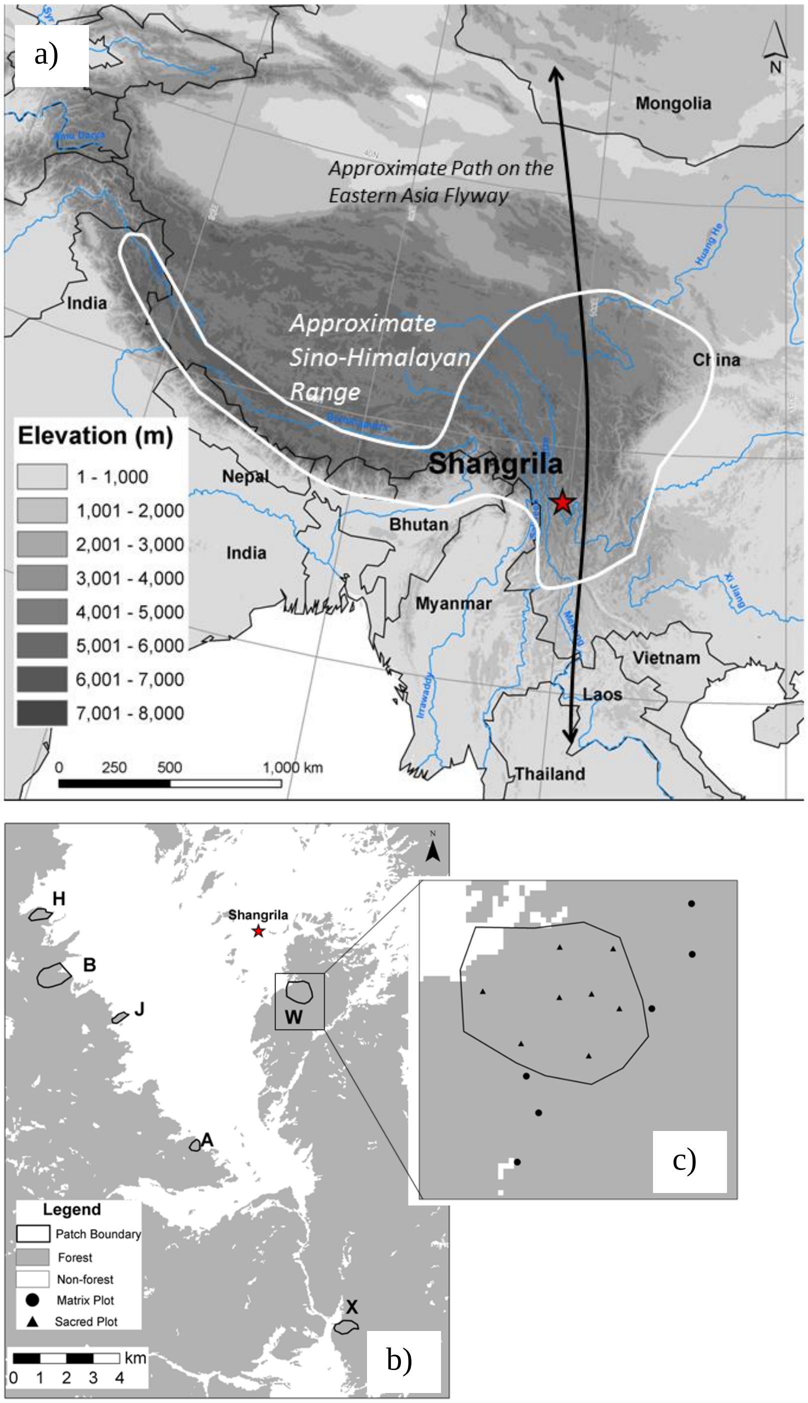


Figure 16. Multivariate analyses of a) habitat gradients (PCA) and b) bird community composition (NMDS).

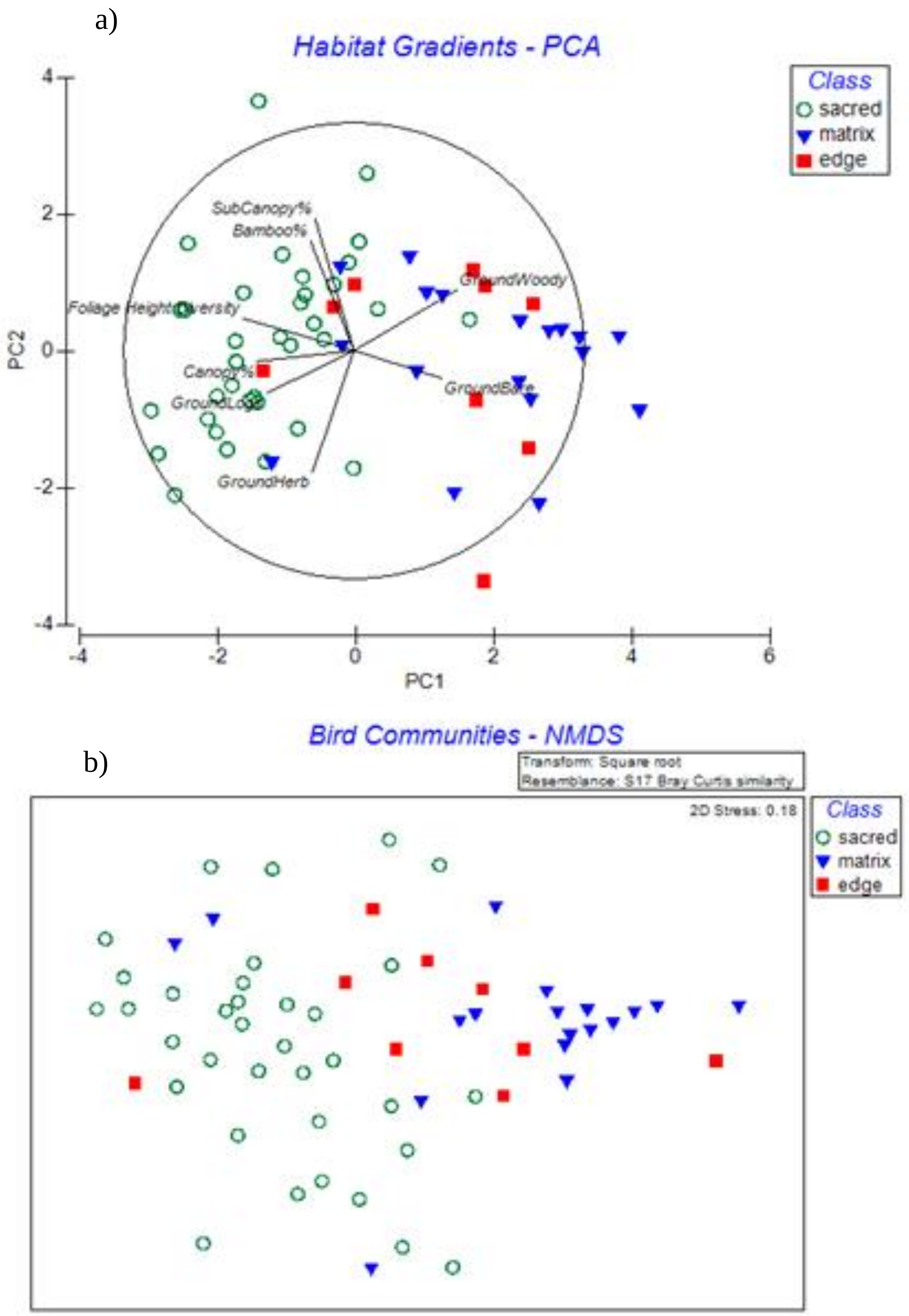


Figure 17. Box plots of a) plot-scale bird species richness, b) plot-scale bird abundance c) patch-scale rarefied bird richness, and d) rarefied species richness accumulation curves at the landscape scale (dotted lines are 95% confidence intervals).

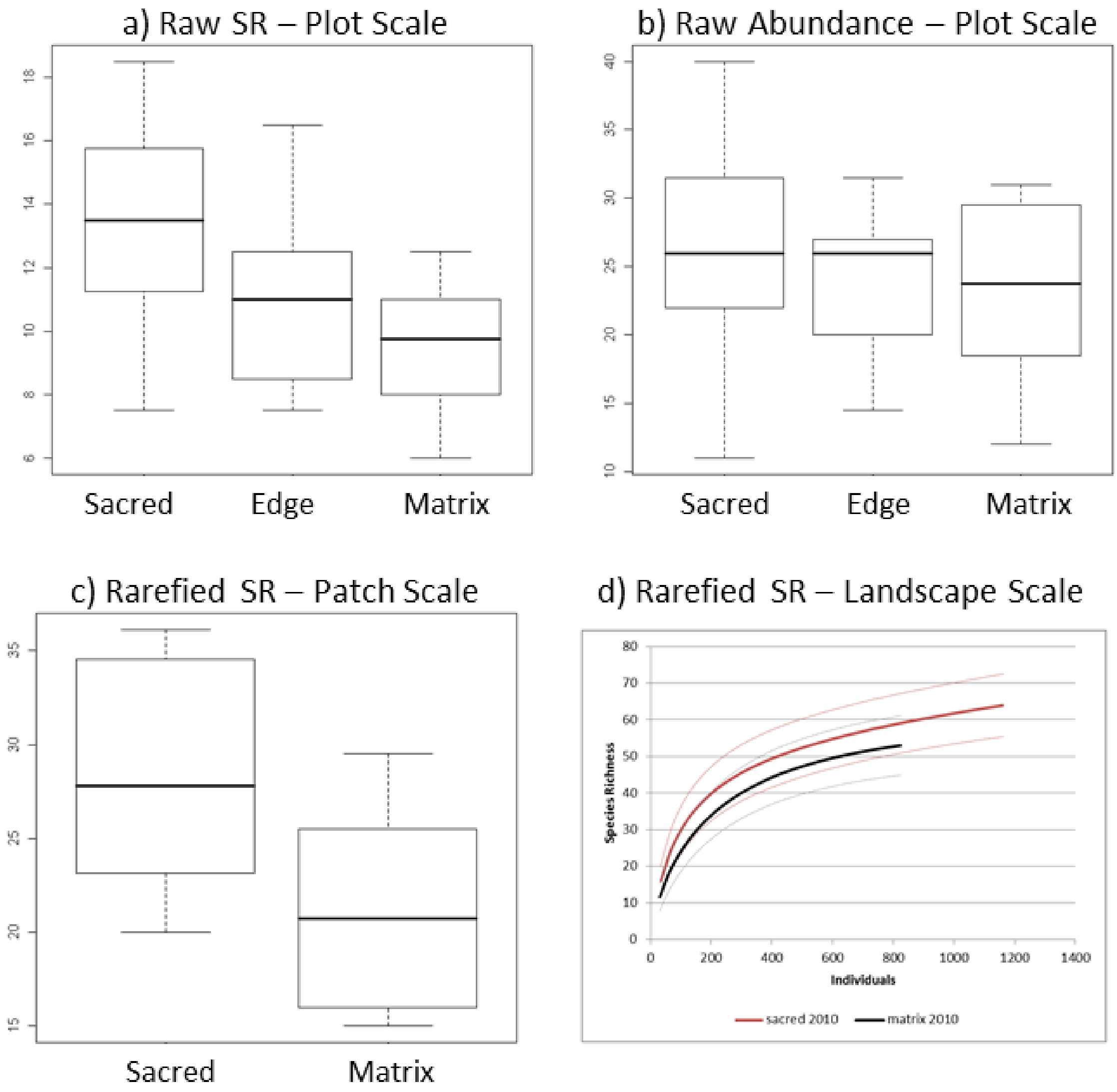
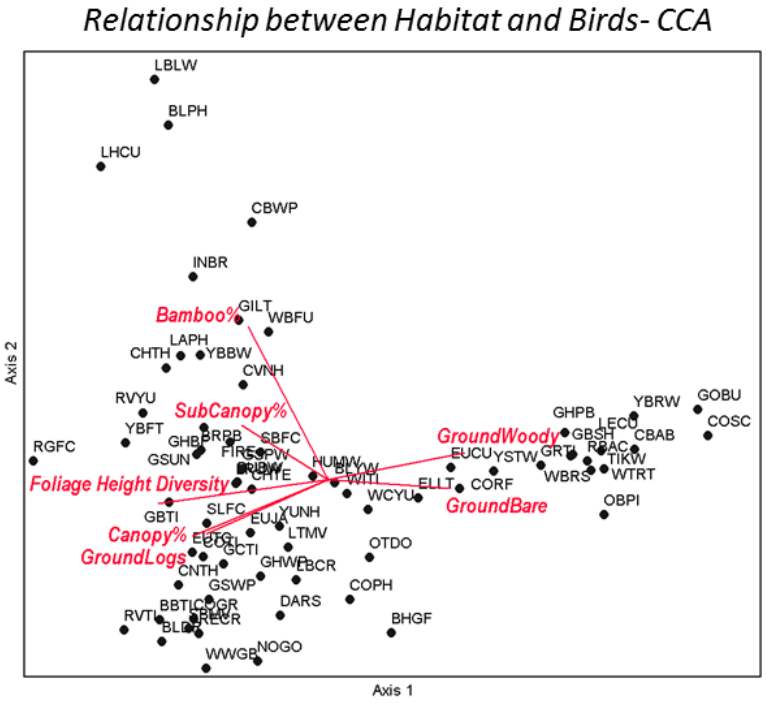


Figure 18. The relationship between habitat gradients and bird communities. Foliage Height Diversity was main characteristic structuring bird communities along Axis 1, and Percent Bamboo and Percent Canopy Cover were the most important variables structuring bird communities along Axis 2.



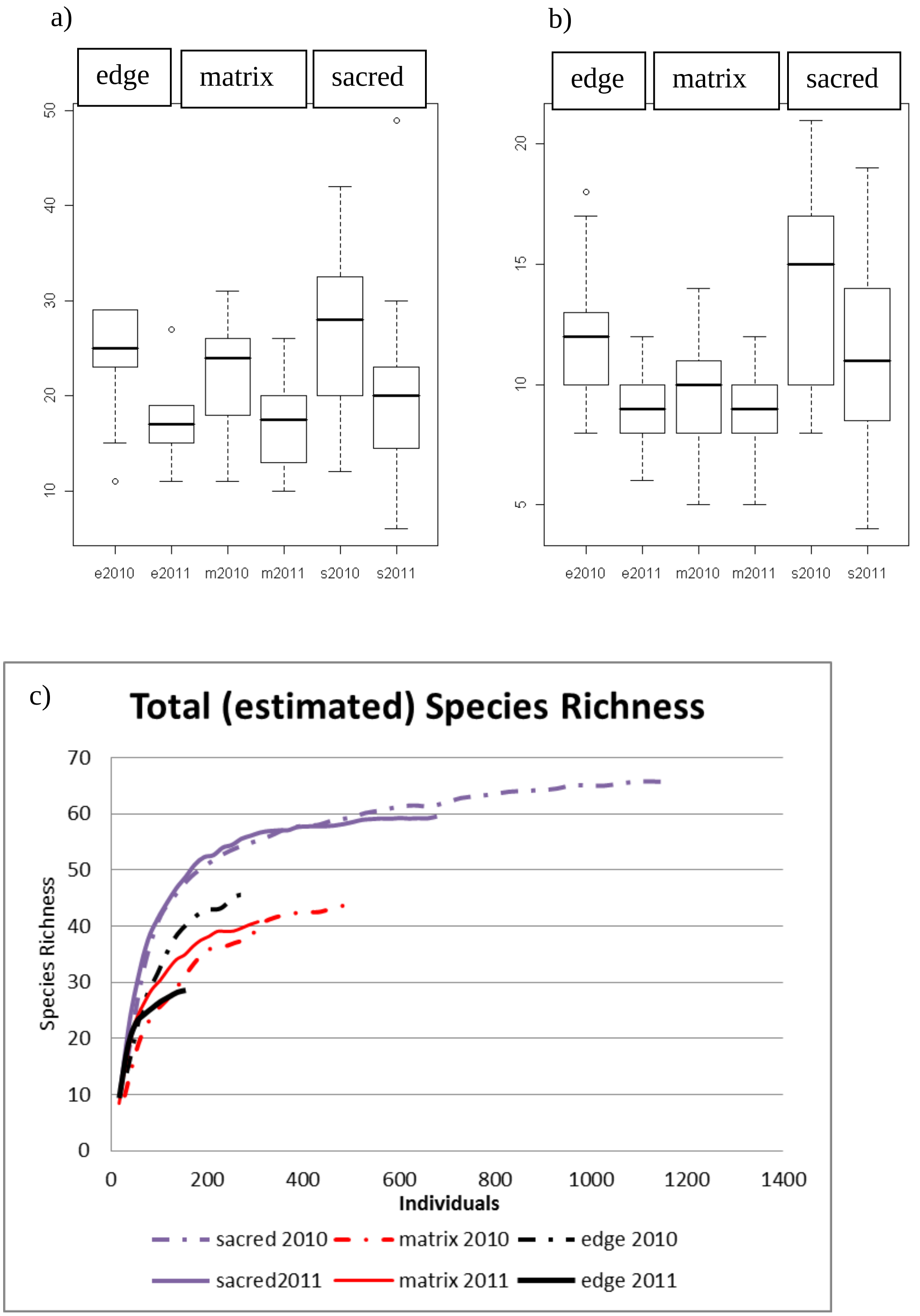


Figure 19. a) Plot-scale abundance, b) Plot-scale species richness and c) landscape-scale total (estimated) species richness in both years

Appendix 1. The set of candidate variables chosen for the stepwise regression, with the corresponding Adjusted R^2 and significance values in the simple linear regressions to predict plot-scale species richness; and means, standard deviations and significance values for each variable in sacred versus matrix habitats.

	Simple Linear Regression Analysis		**Sacred**		**Matrix**		**Difference between Sacred and Matrix**
Variable	Adj. R^2	*p*	Mean	SD	Mean	SD	*p*
Canopy Layer (> 5 m)							
Biggest Tree (dbh)	0.31	< 0.01	63.4	39.7	23.7	24.1	< 0.01
Tallest Tree (m)	0.30	< 0.00	26.7	9.7	11.7	10.4	< 0.01
Foliage Height Diversity	0.13	< 0.01	3.1	0.7	1.8	0.4	< 0.01
Tree Species Richness	0.09	0.01	5.3	1.6	3.7	1.4	< 0.01
Canopy Cover (%)	0.04	0.06	54.9	27.6	12.8	21.7	< 0.01
Basal area (sq. ft./acre)	0.04	0.06	70.1	49.0	15.2	28.6	< 0.01
Sub Canopy Layer (0.5 - 5 m)							
Bamboo Cover (%)	0.20	< 0.01	8.7	12.6	1.9	3.2	0.01
Subcanopy Cover (%)	0.07	0.02	62.4	10.5	52.0	15.2	< 0.01
Sapling Cover (%)	0.05	0.04	20.8	8.5	17.1	12.9	0.181
Ground Layer (0 - 0.5 m)							
Downed Log Cover (%)	0.12	< 0.01	1.6	1.3	0.4	1.0	< 0.01
Bare ground (%)	0.12	< 0.01	0.3	0.6	6.5	7.9	< 0.01
Live Woody Plant Cover (%)	0.05	0.05	39.7	12.0	59.6	13.4	< 0.01

Appendix 2. Description of the 62 plots sampled during the study.

Site	**Class**	**Plot**	**General Habitat Type**	**Dominant Tree Species**	**Foliage Height Diversity**	**Tree Species Richness**	**Shrub Species Richness**
A	Sacred	AA1	Mid-successional forest	*Pinus densata*	2.8	4	10
A	Sacred	AA2	Primary forest	*Quercus aquifolioides*	3.0	7	12
A	Sacred	AA3	Scrub and saplings	*Pinus densata*	2.7	6	12
A	Sacred	AA4	Primary forest	*Quercus aquifolioides*	2.7	7	9
A	Matrix	AT1-260	Scrub forest	na	1.1	4	7
A	Matrix	AT1-510	Scrub forest	na	1.1	2	10
A	Edge	AT1-60	Scrub and saplings	na	1.5	5	11
B	Sacred	BA1	Mid-successional mixed	*Betula platyphylla*	2.7	8	11
B	Sacred	BA2	Primary forest	*Picea likiangensis var. linzhiensis*	2.5	6	13
B	Sacred	BA3	Primary forest	*Picea likiangensis var. linzhiensis*	2.6	6	16
B	Sacred	BA4	Primary forest	*Picea likiangensis var. linzhiensis*	2.4	2	17
B	Sacred	BA5	Primary forest	*Sorbus multijuga*	2.3	3	9
B	Sacred	BA6	Primary forest	*Picea likiangensis var. linzhiensis*	3.5	4	10
B	Sacred	BA7	Primary forest	*Picea likiangensis var. linzhiensis*	3.1	6	12
B	Sacred	BA8	Mid-successional mixed	*Tilia chinensis*	2.4	5	15
B	Sacred	BA9	Mid-successional birch	*Betula platyphylla*	2.8	6	14
B	Matrix	BAT1-260	Scrub forest	na	1.0	2	14
B	Matrix	BAT1-510	Scrub forest	na	1.3	3	17
B	Matrix	BAT2-260	Scrub forest	na	0.7	2	9
B	Matrix	BAT2-510	Scrub forest	na	0.4	4	14
B	Edge	BAT1-60	Scrub and saplings	*Larix potaninii var.macrocarpa*	1.3	5	13
B	Edge	BAT2-60	Scrub and saplings	na	1.7	3	12
H	Sacred	HA1	Early successional birch	*Betula platyphylla*	2.4	6	12
H	Sacred	HA2	Mid-successional mixed	*Picea likiangensis var. linzhiensis*	2.3	9	15
H	Sacred	HA3	Primary forest	*Picea likiangensis var. linzhiensis*	2.7	7	13

H	Sacred	HA4	Primary forest	*Picea likiangensis var. linzhiensis*	3.1	7	12
H	Matrix	HAT1-260	Scrub forest	na	1.5	5	11
H	Matrix	HAT1-510	Scrub forest	na	1.0	2	8
H	Edge	HAT1-60	Scrub and saplings	na	1.4	6	17
J	Sacred	JA1	Mid-successional mixed	*Larix potaninii var.macrocarpa*	1.4	3	10
J	Sacred	JA2	Early successional birch	*Betula platyphylla*	2.8	7	14
J	Sacred	JA3	Early successional birch	*Betula platyphylla*	2.7	5	12
J	Matrix	JAT1-260	Scrub forest	na	0.0	2	8
J	Matrix	JAT1-510	Scrub forest	na	0.5	3	15
J	Edge	JAT1-60	Scrub and meadow	na	0.5	3	12
W	Sacred	WF1	Mid-successional birch	*Betula platyphylla*	2.4	3	14
W	Sacred	WF2	Primary forest	*Larix potaninii var.macrocarpa*	2.7	4	17
W	Sacred	WF3	Primary forest	*Quercus pannosa*	2.8	4	10
W	Sacred	WF4	Primary forest	*Quercus pannosa*	2.8	5	12
W	Sacred	WF5	Primary forest	*Quercus pannosa*	3.1	4	13
W	Sacred	WF6	Primary forest	*Quercus pannosa*	2.8	3	14
W	Sacred	WF7	Primary forest	*Larix potaninii var.macrocarpa*	1.8	4	16
W	Sacred	WF8	Primary forest	*Quercus pannosa*	2.5	4	14
W	Matrix	WAT1-260	Scrub and saplings	*Pinus densata*	1.2	6	10
W	Matrix	WAT1-510	Scrub forest	na	1.0	3	7
W	Matrix	WAT2-260	Scrub forest	na	0.8	4	8
W	Matrix	WAT2-510	Scrub forest	na	0.9	4	9
W	Edge	WAT1-60	Scrub and saplings	*Larix potaninii var.macrocarpa*	1.5	4	8
W	Edge	WAT2-60	Scrub and saplings	*Larix potaninii var.macrocarpa*	2.2	2	15
X	Sacred	XA1	Primary forest	*Picea likiangensis var. linzhiensis*	2.4	6	18
X	Sacred	XA2	Primary forest	*Populus davidiana*	2.2	6	11
X	Sacred	XA3	Primary forest	*Picea likiangensis var. linzhiensis*	2.0	6	11
X	Sacred	XA4	Primary forest	*Salix delavayana*	3.9	5	16
X	Sacred	XA5	Primary forest	*Acer caesium*	2.5	7	12
X	Sacred	XA6	Primary forest	*Acer pectinatum*	2.9	6	9

X	Sacred	XA7	Primary forest	*Salix delavayana*	2.5	6	8
X	Matrix	XAT1-260	Mid-successional mixed	*Betula platyphylla*	2.5	6	18
X	Matrix	XAT1-510	Mid-successional pine	*Pinus densata*	2.3	6	18
X	Matrix	XAT2-260	Early successional mixed	*Pinus densata*	2.2	3	11
X	Matrix	XAT2-510	Early successional pine	*Pinus densata*	2.4	3	9
X	Edge	XAT1-60	Mid-successional pine	*Pinus densata*	2.3	4	7
X	Edge	XAT2-60	Early successional pine	*Pinus densata*	2.7	5	10

Appendix 3. Breeding birds and number of observations during the 2010 and 2011 breeding seasons.

Code	Common name	Latin name	2010	2011	Total
BLYW	Blyth’s Leaf Warbler	*Phylloscopus reguloides*	234	175	409
HUMW	Hume’s Leaf Warbler	*Phylloscopus humei*	196	84	280
CORF	Common Rosefinch	*Carpodacus erythrinus*	166	72	238
GSPW	Golden Spectacled Warbler	*Seicercus burkii*	133	73	206
ELLT	Elliot’s Laughingthrush	*Garrulax elliotii*	136	38	174
LRUW	Lemon-rumped Warbler	*Phylloscopus proregulus*	63	50	113
SLFC	Slaty-backed Flycatcher	*Ficedula hodgsonii*	65	38	103
WBRS	White-bellied Redstart	*Hodgsonius phaenicuroides*	57	40	97
YSTW	Yellow-streaked Warbler	*Phylloscopus armandii*	30	51	81
BUBW	Buff-barred Warbler	*Phylloscopus pulcher*	54	25	79
TIKW	Tickell's Leaf Warbler	*Phylloscopus affinis*	40	26	66
WCYU	White-collared Yuhina	*Yuhina diademata*	36	27	63
GHBF	Grey-headed Bullfinch	*Pyrrhula erythaca*	37	25	62
GSUN	Mrs Gould's Sunbird	*Aethopyga gouldiae*	37	23	60
OTDO	Oriental Turtle Dove	*Streptopelia orientalis*	34	23	57
SBFC	Slaty Blue Flycatcher	*Ficedula tricolor*	37	20	57
GCTI	Grey-crested Tit	*Pyrrhula erythaca*	34	22	56
COTI	Coal Tit	*Parus ater*	28	26	54
YBBW	Yellowish-bellied Bush Warbler	*Cettia acanthizoides*	34	18	52
BBTI	Black-browed Tit	*Aegithalos bonvaloti*	27	17	44
WTRT	White-tailed Rubythroat	*Luscinia pectoralis*	29	14	43
CHTH	Chinese Thrush	*Turdus mupinensis*	33	7	40
LTMV	Long-tailed Minivet	*Pericrocotus ethologus*	24	16	40
RVTI	Rufous-vented Tit	*Periparus rubidiventris*	28	12	40
WBFU	White-browed Fulvetta	*Alcippe vinipectus*	21	16	37
GILT	Giant Laughingthrush	*Garrulax maximus*	26	8	34
GBSH	Grey-backed Shrike	*Lanius tephronotus*	20	10	30
INBR	Indian Blue Robin	*Luscinia brunnea*	19	11	30
CNTH	Chestnut Thrush	*Turdus rubrocanus*	19	5	24
LBLW	Large-billed Leaf Warbler	*Phylloscopus magnirostris*	17	7	24
EUCU	Common Cuckoo	*Cuculus canorus*	6	17	23
DARS	Daurian Redstart	*Phoenicurus auroreus*	14	8	22
BLPH	Blood Pheasant	*Ithaginis cruentus*	12	9	21
WITI	Willow Tit	*Parus montanus*	12	9	21

YUNH	Yunnan Nuthatch	*Sitta yunnanensis*	13	8	21
CHTE	Chestnut-headed Tesia	*Tesia castaneocoronata*	10	8	18
GRTI	Great Tit	*Parus major*	15	3	18
COSC	Common Stonechat	*Saxicola torquata*	12	5	17
CVNH	Chestnut-vented Nuthatch	*Sitta nagaensis*	12	4	16
LBCR	Large-billed Crow (tibetosinensis)	*Corvus macrorhynchos tibetosinensis*	13	3	16
GRNW	Greenish Warbler	*Phylloscopus trochiloides*	15	0	15
OBPI	Olive Backed Pipit	*Anthus hodgsoni*	9	6	15
GCMV	Grey-chinned Minivet	*Pericrocotus solaris*	14	0	14
GSWP	Great Spotted Woodpecker	*Dendrocopos major*	7	6	13
RVYU	Rufous-vented Yuhina	*Yuhina occipitalis*	4	9	13
WBRF	White-browed Rosefinch	*Carpodacus thura*	12	0	12
COPH	Common Pheasant	*Phasianus colchicus*	7	3	10
RECR	Red Crossbill	*Loxia curvirostra*	0	10	10
COGR	Collared Grosbeak	*Mycerobas affinis*	6	3	9
WWGB	White-winged Grosbeak	*Mycerobas carnipes*	7	2	9
BFRS	Blue-fronted Redstart	*Phoenicurus frontalis*	4	4	8
BTTC	Bar-tailed Treecreeper	*Certhia himalayana*	8	0	8
GBTI	Green-backed Tit	*Parus monticolus*	4	4	8
EUTC	Eurasian Treecreeper	*Certhia familiaris*	0	7	7
GHPB	Grey-headed Parrotbill	*Paradoxornis gularis*	3	4	7
CBAB	Chinese Babax	*Babax lanceolatus*	3	3	6
LAPH	Lady Amherst’s Pheasant	*Chrysolophus amherstiae*	5	1	6
RBAC	Rufous-breasted Accentor	*Prunella strophiata*	3	3	6
YBFT	Yellow-bellied Fantail	*Rhipidura hypoxantha*	3	3	6
BRPB	Brown Parrotbill	*Paradoxornis unicolor*	3	2	5
LECU	Lesser Cuckoo	*Cuculus poliocephalus*	0	5	5
EUJA	Eurasian Jay	*Garrulus glandarius*	2	2	4
BWWB	Bar-winged Wren-babbler	*Spelaeornis troglodytoides*	3	0	3
FIRE	Firethroat	*Luscinia pectardens*	1	2	3
GHWP	Grey-headed Woodpecker	*Picus canus*	2	1	3
GOBU	Godlewski’s Bunting	*Emberiza godlewskii*	2	1	3
VIRF	Vinaceous Rosefinch	*Carpodacus vinaceus*	3	0	3
DBRF	Dark-browed Rosefinch	*Carpodacus nipalensis*	2	0	2
RCFU	Rusty-capped Fulvetta	*Schoeniparus dubia*	2	0	2
RGFC	Rufous-gorgeted Flycatcher	*Ficedula strophiata*	0	2	2
SBMV	Short-billed Minivet	*Pericrocotus brevirostris*	0	2	2
BLDR	Black Drongo	*Dicrurus macrocercus*	0	1	1
CBWP	Crimson-breasted Woodpecker	*Dendrocopos cathpharius*	0	1	1
GRPB	Great Parrotbill	*Conostoma oemodium*	1	0	1
NOGO	Northern Goshawk	*Accipiter gentilis*	0	1	1
PALW	Palla's Leaf Warbler	*Phylloscopus proregulus*	1	0	1

RTFC	Red-throated Flycatcher	*Ficedula Parva*	1	0	1
SWGB	Spot-winged Grosbeak	*Mycerobas melanozanthos*	1	0	1
WIWR	Winter Wren	*Troglodytes troglodytes*	1	0	1
WTRO	White-tailed Robin	*Myiomela leucura*	1	0	1
YBRW	Yellow-browed Warbler	*Phylloscopus inornatus*	0	1	1