LAND USE AND LAND COVER CHANGE IN SOUTHWEST CHINA'S HIMALAYAN MOUNTAINS: IMPLICATIONS FOR ALPINE MEADOWS, FOREST ECOSYSTEMS, AND AVIAN BIODIVERSITY

A dissertation proposal submitted by
Jodi S. Brandt
Department of Forest and Wildlife Ecology
University of Wisconsin-Madison
jsbrandt@wisc.edu

Advisor
Volker C. Radeloff
Department of Forest and Wildlife Ecology
University of Wisconsin-Madison

Committee
Mutlu Ozdogan
Department of Forest and Wildlife Ecology
University of Wisconsin-Madison

Anna Pidgeon
Department of Forest and Wildlife Ecology
University of Wisconsin-Madison

Philip Townsend
Department of Forest and Wildlife Ecology
University of Wisconsin-Madison

A-Xing Zhu
Department of Geography
University of Wisconsin-Madison

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OVERVIEW

Biodiversity loss is a global crisis because it profoundly alters ecological processes and disrupts ecosystem services. The rate of species' extinctions has increased exponentially as humans increasingly dominate the majority of the world's natural resources (Chapin et al. 2000). 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 LULCC and biodiversity are not well understood.

For example, simplistic explanations of the drivers of LULCC, such as poverty and population growth, 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. The relationship between LULCC and biodiversity is also complex. The species-area relationship derived from island biogeography (MacArthur and Wilson 1967) asserts that the larger an area, the higher species diversity it will support. However, there are "hotspots", or regions that contain disproportionately high levels of biodiversity (Myers et al. 2000). And it is not simply the amount of habitat that is important for species distribution and viability, but also the distribution of the habitat on the landscape (Turner et al. 2001).

Even the results of our conservation activities are unpredictable. The protected area network has expanded around the globe, but in some cases protected areas are nothing more than "paper parks", due to lack of resources for management and enforcement (Joppa et al. 2008). Even though many parks appear to be effective at protecting forests within their boundaries (Bruner et al. 2001), often the protection of resources within the park are counterbalanced with accelerated degradation outside of PA boundaries (Ewers and Rodrigues 2008; Wittemyer et al. 2008).

Satellite imagery and advances in remote sensing analysis provide a tool to study landscape pattern and understand its influence on ecosystem processes and biodiversity at spatial and temporal scales that was previously impossible (Nagendra 2001; Turner et al. 2003). My research will use established and novel remote sensing techniques coupled with field-based measurements to address broad questions of LULCC, biodiversity, and conservation in a highly diverse, rapidly changing, and little-studied region of Southwest China. Nationwide, China is experiencing strong economic development accompanied by massive environmental degradation (Liu and Diamond 2005). Almost all wilderness is gone in densely-populated eastern China, but China's Southwest, a remote and rugged region of the Himalayans, still retains unique ecosystems with remarkable biodiversity. However, these ecosystems are highly threatened, and China's actions in the upcoming years will determine the ultimate fate of the region's biodiversity. In response to environmental degradation, China's government has enacted several policies to protect natural resources in SW China, such as logging and burning bans, afforestation programs, and nature reserve expansion. However, it is largely unclear what has resulted from these policies. My research will characterize the LULCC patterns resulting from these policies, and the consequences of the LULCC for SW China's unique floral and faunal diversity.

My study area is 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 underwent major changes due to national policies aimed at fostering both economic development and environmental protection. These policies have 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 processes of global climate change (Baker and Moseley 2007). The objective of my **first chapter** is to ask: what are the LULCC patterns that result from the diverse array of driving factors acting at multiple spatial scales? Using Landsat imagery, I will classify land use and land cover at 4 different time steps since 1975 to 1) understand regional-scale patterns of LULCC in the past 3 decades, and 2) provide a context for the next 4 chapters, where I focus on specific ecosystems and policies.

In **Chapter 2,** I focus on the alpine ecosystems, which are among the most diverse and unique in the world. Plant diversity in the alpine meadows is remarkably high, and has evolved in concert with seasonal Tibetan yak herding. A Chinese government ban on intentional burning forced Tibetans to alter their traditional herding practices because the ban prevents herders from burning small patches of shrubs to expand meadow areas for grazing. I hypothesize that in the absence of burning, meadows shrink and heterogeneity is reduced, with negative consequences for alpine meadow species richness. I use both remote sensing and field based measurements to address the implications of the burn ban for alpine meadow biodiversity.

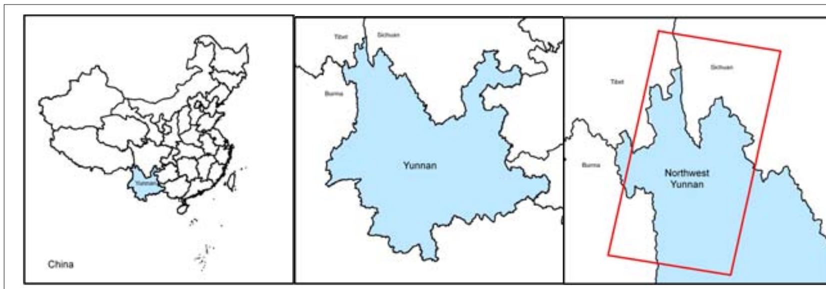


Figure 1. Location of study area in Northwest Yunnan, China. Red lines indicate boundaries of the Landsat images (Path/Row 132/40, 41).

In **Chapter 3**, I direct attention to the highly-threatened temperate forest ecosystems. Chinese statistics report overall afforestation in southwest China since the implementation of a logging ban in 1998. However, reports from the literature and personal observation indicate that natural forests continue to be logged, and the regeneration of forest into a high-diversity ecosystem is unlikely due to soil erosion and compaction, and intensive use by local people. The direct mapping of forest diversity or successional stage is very difficult with Landsat imagery. To overcome this problem, I will explore a promising new trajectory-based analysis technique to identify forest-clearing disturbance for dense time-series of imagery. I will map forest clearing disturbance as a proxy for logging to measure rates and patterns of logging in the past 20 years, to understand local people's response to, and the effectiveness of, the 1998 logging ban.

In **Chapter 4**, I will address the question of protected area effectiveness. Since 1980, five protected areas have been established in the study area, and four more are planned for the next decade. Using my remote sensing analyses from Chapters 1 and 3, I will measure the ability of the existing protected areas to protect forests within their boundaries, and identify areas of leakage outside of their boundaries. I will also ask whether government or traditional forms of protection have been more effective by comparing state-sponsored protected areas with Tibetan sacred forests.

Chapter 5 will further explore the role of Tibetan sacred forests for biodiversity conservation in NW Yunnan. Although the study area is within a center of avian endemism in China (Lei et al. 2003), no research has been done on forest songbird distribution, diversity, population trends, or habitat selection. I will study forest songbird communities in sacred forest patches and the surrounding secondary forest matrix in order to 1) determine habitat selection of the dominant forest songbird species, and 2) investigate whether sacred forests maintain songbird biodiversity in their surrounding landscapes.

To summarize, my research will address broad themes of LULCC, biodiversity, and conservation, with NW Yunnan as my case study. Several characteristics distinguish NW Yunnan from other developing regions, thus making an important contribution to the existing LULCC science and conservation biology literature. First, NW Yunnan harbors mountain temperate forest and alpine ecosystems of a kind not found in Africa or Latin America, where most research is conducted. Second, unlike most developing countries, China has a strong central government and ability to effect change through policy and regulation. Third, in the past 20 years China has implemented several strong environmental policies and numerous nature reserves. Enough time has passed to understand the on-the-ground results of these policies and protected areas. Finally, the region is experiencing accelerated climate change (Baker and Moseley 2007), allowing us an opportunity to gain insights to future dynamics in other regions. Thus, NW Yunnan represents a unique opportunity to study linkages between LULCC and biodiversity in rapidly developing region that is ecologically, socially, and politically unique.

STUDY AREA

NW Yunnan is situated in the northwest corner of Yunnan province (Figure 1), bordering Tibet and Sichuan Province (Figure 2, Landsat TM Path/Row 132/41 and 132/40). NW Yunnan is part of the rugged and inaccessible Hengduan range of the eastern Himalayans. Three rivers (the Yangtze, Mekong, and Salween) create steep gorges, with elevations ranging from 1500 to 6740 m. The climate is temperate and monsoonal, characterized by wet summers (June-September) and dry winters. Average temperature varies widely due to the altitudinal gradient, and annual precipitation averages 350-900mm/year (Baker and Moseley 2007).

The region is a biodiversity hotspot (Myers et al. 2000). Over 7,000 plant species, 410 birds and 170 mammals have been documented in NW Yunnan (Xu and Wilkes 2004; Chang-Le et al. 2007). The region's natural forests are home to endangered Yunnan snub-nosed monkeys and several species of threatened forest pheasants. Yunnan is a center of avian endemism in China (Lei et al. 2003), and high-altitude wetlands and alpine lakes are important migration stops for numerous species of waterbirds, including the endangered black-necked crane (Xu and Wilkes 2004).

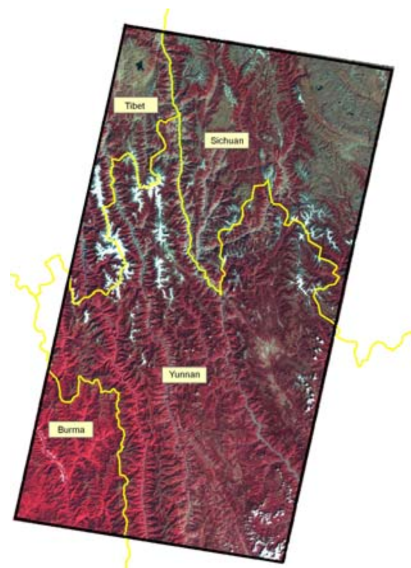


Figure 2. Mosaic of Landsat TM images 132-40 and 132-41 from October 28, 1999.

NW Yunnan is also culturally diverse, and was designated a UNESCO World Heritage Site since 2003. Local people are predominantly Tibetan in ethnicity, culture, and language, with dispersed pockets of other minorities, including Lisu, Bai, Naxi, and Yi peoples. Although the land use practices employed by local people are "traditional", they are not static, and there has been a rapid response and evolution of livelihood practices in the last few decades. Tibetan herders have changed their burning and herding practices in alpine regions due to governmental policies and increased livelihood options (Moseley 2006; Yi et al. 2007). Marginal cropland has been abandoned due to government incentives (Willson 2006; Trac et al. 2007). Local people rely more heavily on livestock and non-timber forest products, increasing the pressure on low-elevation pasture and forest (Yi et al. 2007). In response to the national logging ban, illegal logging of community forests has increased (Zackey 2007). To date, no one has performed a broad-scale analysis of LULCC to determine how these case-study examples manifest themselves in LULCC patterns at the regional scale.

CHAPTER 1

Research question:

What are the dynamics of LULCC in NW Yunnan from 1975 to 2009?

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 underwent major changes. First, China changed from a command economy to a market economy in 1978, spurring economic development throughout the country. The western half of China lagged behind the eastern coastal regions, and in 2000, the federal government initiated a massive "Develop the West" program, which emphasizes infrastructure development, foreign economic investment and education. It has connected formerly remote villages to the market economy, allowing rural farmers to diversify their livelihood strategies, with a decreasing proportion of villagers' income provided by agropastoralism (Yi et al. 2007). Second, in the past decades China has provided incentives for the immigration of culturally-dominant Han Chinese from other parts of China. Such demographic change, along with infrastructure construction, increased pressure on the environment and threatens the balance between people and nature in this historically undeveloped and sparsely populated region. Third, China's government has invested heavily to promote tourism in NW Yunnan. The development of tourism has given local peoples more livelihood options and opportunities, and has resulted in an expansion of the protected area system. Finally, since the 1980s, China has enacted several policies aimed to protect the environment, including a burning ban, a logging ban, and an agricultural conversion program.

It is largely unclear what has resulted from these land use policies. Limited research performed in this region is either at a local scale or only contains anecdotal reports of land cover dynamics. For example, research reports shrub encroachment into the alpine meadows (Baker and Moseley 2007; Melick et al. 2007), cropland abandonment accompanied by failed reforestation efforts (Willson 2006; Trac et al. 2007), intensified grazing pressure on low-elevation pasture and forest (Yi et al. 2007), and illegal logging of community forests (Zackey 2007). I will quantify change at the regional scale by classifying LULCC from satellite imagery in 1975, 1990, 2000, and 2009. Based on these classifications, I will create change maps for three different time periods: 1975-1990, 1990-2000, and 2000-2009. *I predict that the study area has experienced considerable LULCC from 1975 to 2009, and I hypothesize that LULCC trajectories are driven by an interaction of broad-scale factors, including regional economic development, national environmental policies, and topography (H1).* Using a review of the literature, field observations, and conversations with local people and Chinese scientists, I have formed a series of hypotheses of specific LULCC dynamics (Table 1).

<u>Natural Forests:</u> From the 1950s through the 1990s, southwest China's forests were used as a key resource in China's development. Because of its remoteness, NW-Yunnan forests were some of the last to

be exploited. There were massive clearcuts by state logging companies in easily accessible areas (i.e. along roads) during the 1980s (according to conversations in the field). I predict that this deforestation will be detected in the satellite imagery from 1975 to 1990. No information on logging activities during the 1990s is available from the published literature or from field conversations, and the satellite image analysis will provide a record of either deforestation or regrowth from 1990 to 2000. After a disastrous flood of the Yangtze River in 1998, a Logging Ban was implemented to prohibit commercial logging. The ban allows limited logging by villagers for their own use. Therefore, I predict that between 2000 and 2009, cutting of natural forest continued, but at a much lower rate than during previous time periods.

Regeneration of logged areas is of utmost importance for biodiversity conservation in NW Yunnan. Unfortunately, the literature and my field observations indicate that many logged areas are not on a successional trajectory towards natural forests because livestock grazing hampers natural regeneration (Willson 2006; Trac et al. 2007). For example, at lower elevations, homogeneous stands of secondary pine, birch, and oak shrub dominate because they are not palatable for free-grazing cattle. At

Table 1. A summary of the hypothesized change trajectories in NW Yunnan.

Class	**1975-1990**	**1990-2000**	**2000-2009**	**Driving Factor**
Natural Forests	---	--	-	1998 Logging Ban
Agriculture	-	-	--	Regional economic development, migration to cities, 2001 Sloping Land Conversion Program
Shrublands, Secondary Forests, Non-alpine Grassland	+	+	+	Regeneration after logging and agricultural abandonment
Alpine Meadows	-	-	--	1988 Burn Ban
Urban	+	+	+	Migration, Development

higher elevations where climatic conditions limit forest regeneration, and in areas of intensive grazing, grassland remains several years after logging. Based on these observations, I predict that the LULCC analysis will show that logged forests are converted primarily to shrubland, grassland, and homogeneous stands of pine, birch, or oak shrubs. The challenge for the image analysis will be to discriminate between different types of shrublands and forest successional stages in a way that is ecologically meaningful. This topic will be addressed in more detail in Chapter 3.

Cropland: The region is historically populated by rural minorities that practice subsistence-based agropastoralism. I predict that there has been cropland abandonment from 1975 to 1990 and 1990 to 2000 because rural farmers have had increased access to other livelihoods. In 2001, a Sloping Land Conversion Program began, which provides incentives for farmers to convert steep croplands to forest. I predict that this policy resulted in an accelerated rate of cropland abandonment in 2000 to 2009, especially on steep lands. I also predict that abandoned fields have largely been converted to grassland or woody shrubs.

Abandoned fields are often heavily grazed by livestock, inhibiting forest growth (Willson 2006). Although the Sloping Land Conversion Program intends for active reforestation of abandoned fields, the literature indicates that the program in large part has not resulted in successful afforestation (Xu et al. 2004; Trac et al. 2007).

Alpine Meadows: The alpine (>3800 m) region of NW-Yunnan has traditionally been an integral part of the agro-pastoral system. Villagers herd their livestock in the alpine meadows during the summer months, and burn patches of alpine shrub and sub-alpine forest to expand the available pasture. In 1988 a nationwide burning ban was implemented primarily to protect forests, but the ban also effectively prevents herders from burning in the alpine. I predict an increase in forest and shrub cover in the alpine zone at the expense of grassland after the 1988 burn ban. (This topic is addressed in more detail in Chapter 2).

Urban areas: Incentives for agricultural abandonment along with infrastructure and economic development have led to the migration of rural villagers to the cities throughout China in the past decades. According to conversations with locals in the field, this migration is occurring in NW Yunnan, as young people leave the villages to work in the cities. Based on preliminary analysis of satellite images for my study area, I predict that migration and development have resulted in high urban growth around the three provincial capitals in the study area during all three time periods.

Drivers of Change: I will conduct basic analyses to identify broad-scale spatial patterns of land use change and its drivers, including roads, villages, cities, streams and rivers, and topographic variables. For example, I hypothesize that areas close to roads, villages, and cities have suffered greater deforestation, and that natural forests have remained intact in remote and very steep areas. I also hypothesize that steep, high-elevation agricultural fields have been abandoned at a higher rate than flat, low-elevation fields close to rivers. I further hypothesize that although townships and provinces are separate political units with their own governing bodies, that broad-scale drivers (i.e. roads, topography, and government policies) mask the variation in land cover change patterns due to political unit.

A comprehensive identification and quantification of driving factors is beyond the scope of my research, and conclusions drawn from this analysis will be limited due to the very coarse nature of my GIS data. The main objective of this chapter is to characterize broad-scale patterns and drivers of LULCC in the region, which provides a context for in-depth investigation of particular ecosystems, as detailed in the next several chapters.

Technical Approach

I will quantify LULCC at the regional scale by performing a post-classification comparison of independently-classified LULC maps from 1975, 1990, 2000, and 2009. I will use two different classification approaches and test their accuracy. First, I will use a hybrid approach that employs both supervised and unsupervised classification techniques. Second, I will use Support Vector Machines (SVM), a non-parametric machine learning method, to test its relative ability to address multi-class problems in an extremely complex and mountainous environment.

Review of Different Classification Techniques Used in this Study:

Statistical-based supervised and unsupervised classification methods are the most commonly used and well-known methods for satellite image classification. Maximum Likelihood Classification (MLC), a supervised technique, relies on training data to describe the distribution of the spectral data of each class. During classification, each pixel of the image is assigned to the category to which it has the highest statistical probability of membership. Unsupervised classification techniques, such as the ISODATA algorithm, do not use training data. Rather, the user specifies the number of classes desired, and pixels in an image are grouped based on their spectral similarity. Both techniques assume normality when assigning pixels to a class, even though image, land cover, or change detection class data are typically not normally distributed. MLC and unsupervised techniques represent the more conventional, “tried-and-true” method for image classification. Their application has been automated in image analysis software packages for several years, and thus such techniques are widely used. Hybrid techniques of MLC and unsupervised classification have been effective to classify LULCC in this study area (Kuemmerle et al. 2006; Ren et al. 2009) and in another mountainous region (Brandt and Townsend 2006; Kuemmerle et al. 2006).

Non-parametric classifiers are machine learning techniques, and include neural networks, decision trees, and support vector machines (SVM). Non-parametric classifiers use artificial intelligence to recognize patterns in large amounts of data. In the case of land cover classification, each pixel in an image is classified according to the patterns recognized in the training data. Both the quality and quantity of the training data are important. The data must represent the entire range of spectral values for a class, and the training dataset must be large enough so that patterns can be accurately recognized (Huang et al. 2002).

Support vector machines (SVM) is a machine-learning technique that has recently been adapted for multi-class problems, and show much promise (Foody et al. 2006; Kuemmerle et al. 2008; Prischepov unpublished data). SVM employs optimization algorithms to locate boundaries, called hyperplanes, between classes (Huang et al. 2002). The training data at the class boundaries are most important for determining the optimal location of the hyperplanes that separate different classes spectrally (Foody and Mathur 2004). Theoretically, it is more accurate than other non-parametric techniques because SVM works through all of the training data and refines the location of the hyperplane until the optimum boundary is found, unlike other machine-learning techniques which place a cutoff at the first acceptable boundary. The result is a more accurate classification, but the performance gain must be paid for with higher computing needs and longer processing times.

In theory, non-parametric techniques should always be superior to parametric techniques, which are constrained by the assumptions of parametric statistics. However, the performance of any classifier is greatly influenced by the quality of the training data. For example, SVM relies heavily on training data that represents the full range of variability in the image because class boundaries are separated based on the edges of the class. If incorrect data are included in the training set, hyperplanes could be erroneously placed based on a few incorrect points. MLC, on the other hand, may be less affected by inaccurate or unrepresentative training data, since MLC assigns classes based on the majority of the training data.

Image Selection and Processing:

Because of the monsoonal climate, no growing season images are available for the study area. Winter images have been used for the limited remote sensing analysis performed in this region because it is relatively easy to obtain cloud-free images for December through February. However, winter images have many disadvantages, including snow cover, exaggerated illumination effects from topography, and senescent vegetation.

Table 2. Images to be used for the multi-temporal classification analysis. Images from 1974 are from the Landsat MSS sensor.

Time Period	Date
Historical	1/5/1974
1990 Series	11/20/1990
	4/13/1991
2000 Series	10/28/1999
	12/25/2000
	4/13/2000
Current	Fall 2008-09
	Spring 2008-09

I will mitigate difficulties caused by snow cover, seasonality and illumination effects by using multi-temporal Landsat TM/ETM imagery to capture phenology (Wolter et al. 1995). Sporadically, relatively cloud-free fall and spring images are captured by the satellites. I located two historical time periods, circa 1990 and 2000, with pairings of cloud-free images collected in late fall and early spring (Table 2). I plan to use similar dates for the current time period, but this will depend on cloud-free image availability. I will attempt to map the following 10 classes: spruce/fir forest, pine forest, evergreen oak forest, deciduous forest, shrub, grassland, agriculture, bare soil, urban/rock, and water. In addition, I will use Landsat MSS images from January 1974 for an Anderson level 1 classification (Urban, Agriculture, Rangeland, Forest, Water, Barren Land).

Images from the USGS archives are already orthorectified, and images purchased from the Chinese and Thai Ground Stations will be geocorrected to the USGS images using ERDAS IMAGINE Autosync. Cloud and shadow will be masked with eCognition (Baatz et al. 2003). Atmospheric and topographic correction will be performed using a modified 5S-Code (Richter 1998), and the SRTM DEM, gap-filled with 1:200,000 Russian topographic maps (de Ferranti 2008).

I will perform spectral enhancements on the imagery and test their effectiveness to separate LULCC types. For example, tasseled Cap bands 1, 2, and 3 (which emphasize the soil, vegetation, and moisture properties of landscape materials, respectively) are often very useful to discriminate land cover types (Crist and Cicone 1984). Vegetation indices accentuate green vegetation reflectance, and the Soil Adjusted Vegetation Index (SAVI; derived as {LS band 4 – LS band 3} * 1.5/{LS band 4 + LS band 3 + 0.5}) eliminates the effects of soil on the vegetation index (Huete 1989). I will experiment with various layer stacks of the raw bands and spectral enhancements to find the most effective way to capture the phenological information. Topographic information, such as elevation, slope, and aspect, certainly drive vegetation patterns and will also be included in the layer stacks.

Training Data and Accuracy Assessment:

Three different datasets will serve as training and reference data. First, I collected photographs throughout the study area in summer of 2008, and will collect additional data during my 2009 and 2010 field seasons. These photos will be converted into point data for use in training and validation, and represent on-the-ground field knowledge of the vegetation in the study area. Second, approximately 500 points were collected for different forest types by our Chinese collaborators during 2002 and 2004. Much of these data were collected in remote areas or sites with restricted access, including the Baima Nature Reserve. Third, I will digitize points from high-resolution Quickbird images in Google Earth, which cover about 35% of the study area. The collection of abundant and fully representative training data for some classes in my study area (e.g., natural forest classes) will probably not be possible. I do not have permits to access large areas, and it is impossible to determine between forest types on Google Earth imagery. Thus, it is important to note that the training data for forest classes may be smaller and less representative than is ideal for a machine learning method such as SVM.

I will retain parts of each of the three ground truth datasets as independent reference data for accuracy assessment. I will perform an accuracy assessment for the 2009 classification, and produce a confusion matrix that include user, producer, overall and KHAT accuracy measures. Data to test the accuracy of the 1975, 1990 and 2000 classifications are not available. However, many areas, such as cities, lakes, roads, remote forests, and bare mountaintops are unchanged in all years. Agricultural areas, dense forests, and rangeland have distinct geometric shape and/or spectral signatures and are easily discernible in the images. Site knowledge and close comparison of the image and the classification will allow a qualitative accuracy assessment.

GIS data and Analysis of Drivers of LULCC:

I have GIS layers of road networks, villages and town locations, and provincial and township boundaries, all digitized from 1:250,000 topographic maps from the late 1990s. I will analyze these GIS layers with change maps created from three different time periods: 1975-1990, 1990-2000, and 2000-

2009. For example, I will create 100-m buffer zones extending from roads, and quantify deforestation rates in each zone. I hypothesize that the zone closest to the roads will have the highest rate of deforestation, and that deforestation rates will decrease as distance from a road increases. Furthermore, I will interpret the differences in the 3 change maps in terms of the three national polices implemented during the study period. For example, I will test my hypothesis that deforestation rates decreased following the 1998 logging ban by comparing deforestation rates in 1975-1990, 1990-2000, and 2000-2009.

Chapter 1: Expected Outcomes and Significance

Findings of the first chapter will be summarized in a manuscript submitted to Remote Sensing of Environment. It will include a description of the two different classification methods and a comparison of their accuracy. It will also summarize the patterns of change observed over the study period, and an assessment of whether change has been driven by broad-scale drivers. This research is relevant to the wider remote sensing audience because it compares two very different methods of classification. In addition, it has significance to LULCC scientists because it explores regional LULC change and its drivers in a unique part of the world that has not yet been studied.

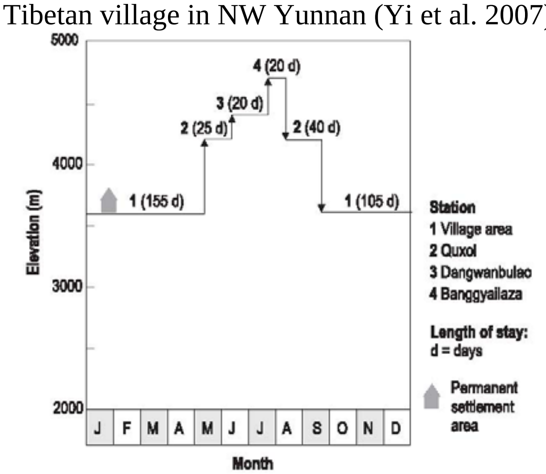
CHAPTER 2

Research question:

Has the burn ban had negative consequences for the diversity of the alpine meadows?

The alpine ecosystems of NW Yunnan are unique, extremely diverse and highly threatened (Xu and Wilkes 2004). The alpine zone is located between ~3800m and 5200m, and consists of three community types, shrub, meadow and scree, that are arranged according to the altitudinal gradient. Shrub, consisting of evergreen and deciduous woody shrub communities, dominate the lower alpine. Meadow, a developed soil substrate covered by grasses, sedges and forbs, is above shrub. Scree, a loose rock substrate with sparse herbaceous vegetation, is above meadow. Above scree is permanent snow and ice at about 5200m. The mean annual temperature above 4100m is less than 0° C (Sherman 2008).

Figure 3. Transhumance herding patterns in a Tibetan village in NW Yunnan (Yi et al. 2007).



The alpine meadows are the most diverse of the three alpine communities, and exhibit remarkable herbaceous species richness (Sherman et al. 2008). Although some cite grazing as a threat to alpine meadow biodiversity (Xu and Wilkes 2004), these ecosystems have evolved in concert with yak husbandry. Native Tibetans practice agro-pastoral transhumance, and have used the alpine as summer pastures for centuries. Herders migrate up to the alpine with their herds in June and return to the village in September, and typically have temporary camps at several elevations (Figure 3). Cattle roam free but return back to the camp every night. Except for a heavily-used ring of degraded pasture around the camps and near water sources, grazing pressure is homogeneous and relatively low-intensity (personal observation). Herders burn small (0.1 – 1 ha) patches of shrubs to expand meadow area. The location and extent of burning is decided by each individual herder depending on his herd size and the shrub/meadow mosaic in his territory.

The ecology of the Chinese Himalayan alpine regions is largely unknown, but it is possible that their remarkable biodiversity has arisen because of traditional Tibetan pastoral management. Regardless, in the past decades there are several major forces of change acting at multiple spatial scales, including global climate change, national environmental protection policies, and economic development, that impact alpine ecosystems in NY Yunnan. *I hypothesize that these forces of change result in a decrease in the heterogeneity of the alpine ecosystems at the regional, landscape, and local scales, with negative consequences for alpine biodiversity (H2).*

<u>Shrub Encroachment and Shrinking Alpine Meadows</u>

One process by which traditional management may serve to create a more diverse plant community than would otherwise exist is that burning increases the spatial extent of meadows. Following from the Species-Area relationship of island biogeography (MacArthur and Wilson 1967), a larger area allows for greater species diversity. Especially in the case of alpine meadow expansion, there is increased opportunity for herbaceous species diversification, because meadows are created at elevations and microclimates where shrubs would otherwise dominate.

A major driver of alpine change during the last 20 years is a ban against intentional burning in the mountains of southwest China. Implemented in 1988 by the Chinese government, the ban intends to protect mountain forests to avoid erosion and flooding, but it also prohibits shrub burning in the alpine. Without burning, woody vegetation increases, and meadow area shrinks. As alpine meadows shrink, grazing pressure on existing meadows rise (Yi et al. 2007). *I hypothesize that shrubs are encroaching into alpine meadows (H2.1).*

During field visits in 2008, I found evidence that the burn ban is variously applied. Field observations and interviews with local herders indicated that adherence to the burn ban depended on the territories' remoteness. In remote regions, where there is little risk that burning will be noticed, traditional practices continue. In alpine areas that are either visible (i.e., along roads) or controlled (i.e., protected areas) burning practices have largely ceased and shrub encroachment is widespread. In addition, some management practices (e.g., the way in which alpine territories are allotted to individual herders) vary depending on the village, and may manifest in different burning practices and vegetation patterns among villages.

The burn ban is not the only broad-scale driver of change in the region. Since the 1980s, economic development in western China has opened up rural areas to a more regional economy, which has led to a decreased use of the alpine meadows for yak grazing in some villages (Yi et al. 2007). Furthermore, nature reserves have been implemented since the early 1980s. They represent a major shift in land tenure rights and have resulted in land use change in areas within the nature reserve boundaries (Harkness 1998; Xu and Wilkes 2004).

Climate change contributes additionally to shrub encroachment. Encroachment started prior to the inception of the burn ban (Li and Walker 1986), and historical climate data and repeat photography indicate climate-induced changes in the alpine since the 1920s. For example, climate data show that temperature has increased at a rate greater than the rest of China. Precipitation trends are more variable, but indicate a drying trend (Baker and Moseley 2007). In addition, all glaciers in the region have receded since the 1920s, tree-line has increased in elevation, and shrubs have encroached into alpine meadows (Moseley 2006).

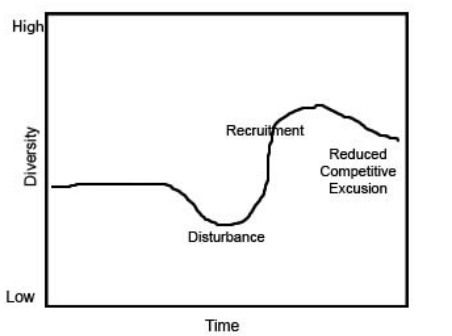
In summary, ample evidence indicates that shrub encroachment and meadow degradation have resulted from a combination of a change in land use practices and climate (Xu et al. 2005; Moseley 2006; Willson 2006; Yi et al. 2007; Sherman et al. 2008). However the extent and patterns of encroachment has not been quantified at the regional scale. *I hypothesize that there is more shrub encroachment closer to roads, in nature reserves, closer to villages, and in areas with less topography (H2.2). By comparing areas with different patterns of change, I will estimate climate-induced shrub encroachment (H2.3).*

<u>The Intermediate Disturbance Hypothesis (IDH)</u>

A second process via which traditional practices may enhance biodiversity is that low-intensity seasonal grazing and infrequent, patchy burning may act as spatially and temporally "intermediate disturbances" that create optimum conditions for species coexistence. Disturbance is the destruction of biomass, which opens up resources for new species. Following disturbance, there is successional recovery of the disturbed area, which is highly driven by the variability in species' life strategies. Immediately following a disturbance, pioneer species will dominate. As time passes, there is a high recruitment of a variety of species with different life strategies. In the absence of disturbance, the strongest competitors will eventually take over. If intermediate temporal disturbance occurs at a frequency that prevents dominance by the pioneer or climax succesional stages, the maximum numbers of species will co-exist (Figure 4).

Although intuitively appealing, experimental research to test the Intermediate Disturbance Hypothesis shows that the hump-shaped response to disturbance is not universal (Hughes et al. 2007). The shape of the diversity curve is highly dependent, among other things, on the transition times between successional stages, the transition type, and the frequency of the disturbance (Johst and Huth 2005).

Figure 4. Trajectory of biodiversity after disturbance as predicted by the Intermediate Disturbance Hypothesis.

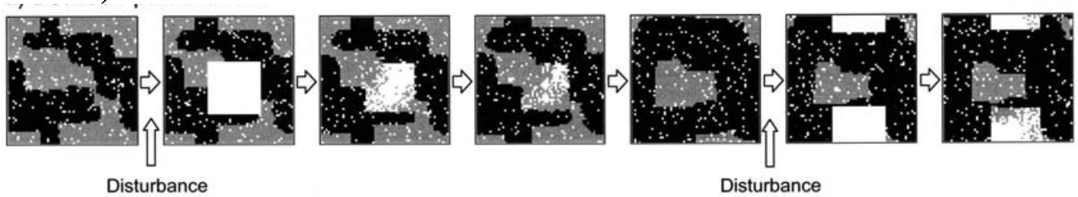


In my study area, burning occurs at a low temporal frequency. This may represent an "intermediate" disturbance that increases plant biodiversity at the patch scale. Annual burning would most likely promote the dominance of pioneer species, such as grasses, forbs and sedges. A complete absence of burning, on the other hand, would promote the dominance of competitive species, such as woody shrubs. Since no information is available regarding post-burn succession in these ecosystems, it is unknown if the intermediate post-burn succession stages provide the optimum conditions for species coexistence. *I hypothesize that the post-burn successional trajectory follows the hump-shaped biodiversity curve predicted by the Intermediate Disturbance Hypothesis (H2.4).*

<u>Intermediate Disturbance Hypothesis – Patchy disturbance</u>

The above scenario (Figure 4) describes the successional trajectory within a disturbed patch after a spatially *homogeneous* disturbance. When a disturbance is spatially *irregular*, patches of different successional stages result in a more spatially heterogeneous landscape (Figure 5). Since within-patch succession is additionally influenced by neighborhood effects of the matrix vegetation (Johst and Huth 2005), a more spatially heterogeneous landscape may result in more variable and complex within-patch succession.

Figure 5. This hypothesized succession after patchy disturbance shows a landscape mosaic of a strong colonizer (e.g. grass, in gray), a strong competitor (e.g. shrub, in black) and post-disturbance (e.g. bare ground, in white). Immediately after the disturbance, the bare ground is quickly colonized by grasses. As time goes on, the disturbed area is completely covered in vegetation and the shrub encroaches into patch. Prior to the system being dominated by shrub, another disturbance patch occurs. If the disturbance occurs very frequently, only grass will persist, and if it occurs very infrequently, shrub will dominate (Roxburgh et al. 2004).



Spatial heterogeneity creates higher biodiversity

Habitat heterogeneity is essential to maintain biodiversity at broad temporal and spatial scales (Belsky and Canham 1994; Bennett et al. 2006; Parr and Andersen 2006). Although we typically view human land use as negative for biodiversity, there is evidence that traditional management systems can serve as a patchy and diverse disturbance regime that creates semi-natural habitats with high spatial heterogeneity, resulting in excellent conditions for species coexistence (Vandvik et al. 2005). For example, the European Alps were historically a heterogeneous mosaic of crop fields, meadows, shrubs and woodlands when used for small-scale agropastoralism (Chauchard et al. 2007). With land abandonment, homogenization has occurred at a regional scale, with negative impacts for biodiversity (Laiolo et al. 2004; Anthelme et al. 2007; Chauchard et al. 2007). Norwegian heath ecosystems also decrease in spatial heterogeneity when traditional forms of pastoralism are abandoned (Vandvik et al. 2005). Since European settlement, tallgrass savanna ecosystems have homogenized due to the contemporary pasture management and nature reserves, resulting in reduced plant and avian biodiversity (Fuhlendorf et al. 2006).

The alteration in spatial heterogeneity on the landscape due to the introduction or withdrawal of a disturbance depends on the environmental context and the disturbance regime in which the ecosystem has evolved (Adler et al. 2001; Parr and Andersen 2006). Grazing and fire are the two major “disturbances” in the Tibetan alpine landscape. Both can alter the spatial heterogeneity of vegetation and they interact in important ways. For example, recently burned patches with sufficient regrowth are preferred by grazers, which allows biomass to build up on unburned patches, increasing their vulnerability to fire (Fuhlendorf et al. 2006). These interactions can drive complex processes that in many cases enhance habitat heterogeneity and species diversity (Vandvik et al. 2005).

In addition to contributing to the loss in the spatial extent of meadows, the burn ban further threatens biodiversity by reducing the spatial heterogeneity of the alpine ecosystems. In this region, the vegetation patterns will be naturally heterogeneous due to the topographic and microclimatic variability. However, traditional management may increase heterogeneity because it introduces patches of vegetation in various stages of post-burn succession. *I hypothesize that patchy burning increases spatial heterogeneity of the alpine ecosystem at the landscape scale (H2.5).*

Technical Approach

I will address aspects of alpine biodiversity at three spatial scales. First, at the regional scale, I will use Landsat images to determine the broad-scale change. I will look at the patterns of change in relation to other aspects of the landscape (roads, nature reserves, and villages) to determine and distinguish the different drivers of shrub encroachment. Second, I will investigate the successional stages after burning, in order to determine whether this practice may enhance biodiversity at the patch scale. Finally, at the landscape scale, I will measure habitat heterogeneity, in order to understand how patchiness at a particular site may vary with changes in land use practices.

This research is being performed in collaboration with Michelle Haynes, a PhD student in the botany department, who is also part of the Yunnan IGERT team. Her other research goals are to 1) assess meadow and shrub communities' responses to different grazing pressures (high-intensity and ambient) and 2) determine how functional trait distributions vary in relation to grazing intensity. Michelle's several years of experience are essential for plant identification and her ongoing research activities will help to guide site selection.

2.1 Spectral Mixture Analysis to quantify shrub encroachment at the regional scale

What is the extent of shrub encroachment since 1990?

A semi-quantitative assessment of encroachment, using aerial photos taken 100 years apart, found that at Baima Snow Mountain, treeline advanced 67 m in elevation and 270 m vertically since 1923. Furthermore, in this same location, tree-line species composition also changed, from spruce and fir to native deciduous larch (normally found at lower elevations) and rhododendron (normally found in the forest understory).

It is unknown whether shrub encroachment has occurred since 1975 to an extent that is measurable with Landsat imagery. I will attempt to quantify shrub encroachment using two techniques. First, I will use post-classification comparisons (derived from Chapter 1) to detect LULCC for 1975, 1990, 2000, and the present. In addition, I will use Spectral Mixture Analysis (SMA) to quantify the amount of green vegetation, non-photosynthetic vegetation, and bare soil within each pixel. SMA is especially useful to detect subtle change within a given land cover class (Brandt and Townsend 2006).

I will apply SMA to the autumn (snow-free) images in order to discern evergreen shrubs from senesced herbaceous vegetation. In the field I observed that, overall, evergreen shrubs make up the majority of total shrub coverage (~90%). Evergreen shrubs include broadleaf *Rhododendron spp*. and *Ericaceae spp.*, and coniferous *Juniperus spp.* Deciduous shrubs make up a minor proportion of the total shrub coverage (~10%), and are almost always closely associated with an evergreen species. Common deciduous species observed in the field include *Potentilla spp.* and *Berberis spp.* It is likely that spectral confusion between the senescent deciduous shrubs, senescent herbaceous vegetation and bare soil will occur in the late autumn images. However, the evergreen shrubs, which represent the majority of woody shrubs in the alpine, should be accurately estimated by the Green Vegetation endmember of the spectral mixture analysis. I will measure the change in proportion of GV on a per pixel basis to detect shrub increase in areas already invaded since 1990.

In addition to the image processing described in chapter 1, I will spectrally normalize the images used for the SMA via Relative Normalization (Collins and Woodcock 1996). Spectral normalization is necessary when directly comparing pixel values of different images to account for differences in atmospheric conditions, sensor variation, or other factors. I will select image endmembers of NPV, GV, bare soil and shade from the 2008 autumn image using the pixel purity index (PPI), field data, and analysis of spectral signatures, and use these endmembers for all image dates.

Accuracy of the results of the 2008, 2000 and 1990 SMA will be assessed by examining the root-mean-square error images, and by looking at the outputs of the mixing model to show that 1) patterns of fraction images coincide with known temporally-invariant areas, 2) endmember fractions are between 0 and 1, and 3) endmember fractions for a pixel sum to 1.

I will additionally assess the accuracy of the 2008 fraction images with field data. In fall of 2008, we collected detailed cover type data at 30-cm intervals along 100-m line-point-intercept transects at 7 sites (Herrick et al. 2005) to quantitatively assess the proportion of GV, NPV, and bare soil. At these sites, we also performed a semi-quantitative estimation of woody shrub, herbaceous and bare ground cover in 50-m swaths extending perpendicular from the transect, to assess the ability of the estimation method to accurately characterize land cover proportions. We used the same semi-quantitative estimation method in 100x100-m quadrats at 14 additional sites.

I will also investigate the infilling of the tree-line ecocline by larch that has been reported (Baker and Moseley 2007) and that I observed in the field (2008). Since larch has a very bright yellow senescent stage in late October-early November, it is possible that one or more of the spectral bands of the autumn imagery can be used to discriminate larch from other tree-line species (generally evergreen species of fir, spruce, oak and rhododendron).

2.2 Determining spatial patterns of alpine change.

Do patterns of shrub encroachment vary on the landscape, and what are the causes of that variation?

The recent changes in land use practices are not uniform throughout the region. In 2008, we visited five alpine areas with similar topography but different management systems (Table 3). I hypothesize that patterns of shrub encroachment will be variable. For example, patch burning was much more evident in remote or non-PA areas, where there would be little danger of enforcement. Thus, I predict that shrub cover will be greater within nature reserves versus outside of nature reserves.

Table 3. Location and burning practices of alpine territories visited in 2008.

Site Name	Protected area?	Near a road?	Current Burning Practices
AP1	No	No	Traditional
AP2	Yes	Yes (primary)	None
AP3	Yes	No	Limited
AP4	No	Yes (secondary)	Limited
AP5	No	No	Traditional

Due to the remote, mountainous topography, not all villages and territories are exposed to the same outside influences or economic opportunities. Economic development has led to abandonment of alpine grazing in some villages, while the pastoral practices in remote villages remain unchanged (Yi et al. 2007). In recent years, construction of roads has concentrated summer herder huts and livestock along roads (Yi et al. 2007), while at the same time making them very visible in terms of burn ban enforcement. I predict that shrub cover will decrease as distance from roads increases. I also hypothesize that grazing pressure increases near roads, but this may be impossible to detect with non-growing season imagery.

I also predict patterns of encroachment with topography (slope, elevation, aspect), political boundaries (township), or population centers (distance of alpine to a village, distance of village to a road). I will test the significance of all of these factors with shrubland encroachment using GIS data, including a DEM raster, and shapefiles of the road networks, population centers, and nature reserves.

*2.3 **Separating climate versus burn ban-induced encroachment***

What proportion of shrub encroachment is due to global climate change?

There was measurable shrub encroachment between 1923 and 2003, at least part of which can be attributed to a warming climate (Baker and Moseley 2007). Since their study used only two points in time, we do not know temporal patterns of change, such as the annual rate of encroachment. My research will quantify shrub encroachment over a much shorter time period with a denser temporal resolution.

Since climate-induced change typically manifests at longer temporal scales, it is possible that the encroachment observed during my study period is all or mostly due to more immediate changes in land use.

I will attempt to determine the relative contribution of climate change to shrub encroachment using the relationships found in Chapter 2.2. For example, I will isolate within-territory gradients of shrub encroachment due to road or village proximity, and then apply a piosphere concept or gradient approach (Pickup and Chewings 1994; Roder et al. 2007), to tease out the relative influence of climate change and land use practices on shrub encroachment. I predict, for example, that within-territory gradients in AP4 (i.e. shrub encroachment decreases as distance from the road increases) are due to different levels of enforcement, whereas within-territory patterns in AP1 and AP5 should be a product of climate change, since all parts of these territories are quite remote.

2.4 Chronosequence approach to determine post-burn successional trajectories

What are the effects of patch burning on plant species composition, and how long do these effects persist?

I hypothesize that the infrequent, patchy fires and minimal grazing of Tibetan traditional herding practices create optimal conditions for species coexistence, based on the Intermediate Disturbance Hypothesis. In order to test this hypothesis, it is essential to investigate the composition and life-span of the post-disturbance successional stages. I will use a chronosequence approach, or "space-for-time substitution" (Kost and De Steven 2000; Bond-Lamberty et al. 2003; von Oheimb et al. 2007; Benscoter 2008; Gonzalez-Tagle et al. 2008), to determine post-burn successional stages in the alpine meadow-shrub mosaic in site AP1. We (assisted by local herders) will identify as many post-burn patches as possible. We will pair each burn patch with an adjacent control or "undisturbed" patch (no physical or anecdotal evidence of burning) that is similar in environmental characteristics (topography, slope, soil type) as the burn patch. We will record the time since burn, according to the herders. If live shrubs are present within the patch, we will collect discs for dendrochronological analysis to verify the approximate age since last burn.

Our study design, i.e., the number and the burn history of the patches, will be limited by conditions that we encounter in the field. We have chosen AP1 as our site because 1) traditional burning practices persist and 2) we have established a working relationship with a local herder who acts as our guide. We will be restricted to sampling only those patches to which we are allowed access, and for which our guide knows the burn history. In general, we will attempt to capture three post-burn successional stages: early (2-3 years), Middle (8-9 years) and late (14-15 years). If possible, we will attempt to collect multiple patches (e.g. 3 or more) for each successional stage to evaluate within-stage variability. We will also attempt to collect single patches between stages (e.g. 5 years, 12 years after burn), in order to increase the temporal resolution of the successional trajectory.

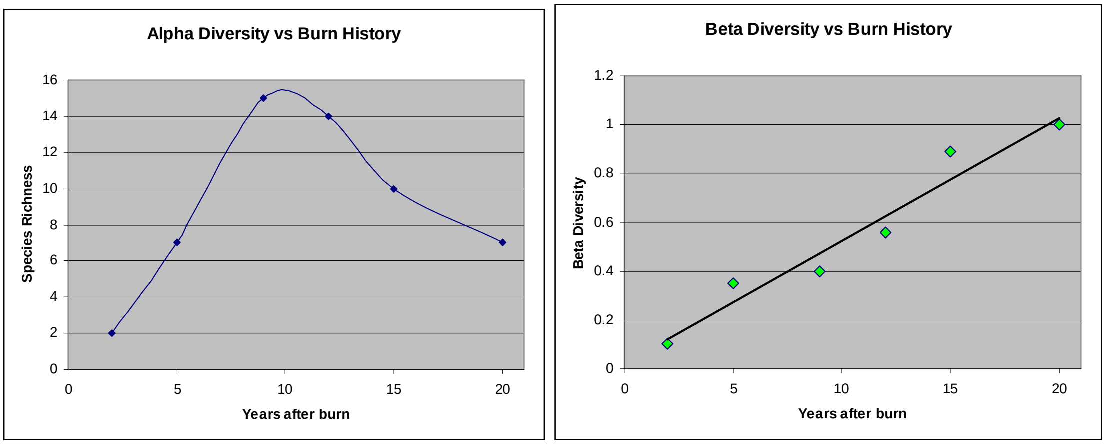
Where possible, we will collect patch data in other alpine territories to booster sample size, evaluate successional trends in other areas, and perform among-site comparisons. For example, sites AP2 and AP4 are accessible by car, and species composition data could be collected at various sites. However, we may not encounter any herders willing to impart knowledge about burning history at these sites.

We will collect patch data in 25 quadrat plots (1m x 1m) within each patch. Quadrat plots will be placed systematically by a) locating a primary transect up slope for the total length of the patch, then, b) place 5 evenly-spaced secondary transects perpendicular to the primary transect, and finally, c) place 5 evenly-spaced quadrats on each of the secondary transects. Within each quadrat we will record the species present and height of vegetation. At each secondary transect we will record distance from nearest herding hut, distance from nearest stream, altitude, aspect, slope, and slope position. At each patch we will record grazing intensity (number of livestock/number of days) and burn history according to the herder.

Data from the 25 quadrat plots will be combined so that we have patch-scale measures of species composition for each patch. We will use several different measures to test the effect of patch burning on biodiversity. Alpha diversity, or within-patch diversity, will be measured by species richness, species evenness, and the Simpson index. Beta diversity, or between-patch diversity, will be measured by the Sorenson index. Gamma diversity is the total of alpha and beta diversity, and represents landscape-scale diversity.

To determine the post-burn pattern of biodiversity, we will plot alpha diversity at each successional stage. I hypothesize that the curve will adhere to that predicted by the IDH (Figure 6). We will also perform paired comparison tests to determine whether the alpha diversity of burn patches is significantly different than alpha diversity of the control patches. I will plot beta diversity at each successional stage to determine how species composition of a burned and its adjacent undisturbed pair vary through time. This is important in terms of dispersal, colonization, and encroachment. I hypothesize a positive linear relationship between burn history and beta diversity, where immediate post-burn herbaceous communities are completely different from the mature community, and as time passes, the disturbed site gradually succeeds to a mature state (Figure 6). Finally, we will determine whether burn patches increase the plant diversity on the landscape scale by comparing gamma diversity of all burned versus all unburned plots. We hypothesize that burned patches have a higher gamma diversity than the control patches.

Figure 6. The graph on the left represents hypothesized changes in Alpha Diversity between burned and unburned pairs at different points in post-fire succession. On the right, hypothesized changes in Beta Diversity between burned and unburned pairs at different points in post-fire succession. Beta is a measure of diversity between two sites. A beta of zero indicates that there are no species in common and a beta of 1 indicates that both sites share all species. The 20-year point in both graphs represent the undisturbed control plots.



2.5 Analysis of high-resolution photographs to measure landscape heterogeneity

What is the habitat heterogeneity within each site, and how does this relate to land use practices?

Spatial heterogeneity of vegetation at multiple scales is altered by human-induced disturbances, creating habitat heterogeneity (Bar Massada et al. 2008). In turn, differences in habitat heterogeneity effects biodiversity at higher trophic levels (St-Louis et al. 2006). I will measure habitat heterogeneity at two sites with similar environmental conditions but different management practices (Table 1). AP1 is a remote territory where traditional burning practices and grazing practices have remained stable over the past decades. Site AP2 is within the Baima Nature Reserve and near the main highway to Tibet. Burning is now non-existent, but grazing pressure appears to be more intensive than other alpine areas. My hypothesis is that AP1 is “patchier” at the landscape scale; i.e. spatial heterogeneity is enhanced due to the patchy burning and minimal grazing pressure.

It is important to quantify spatial heterogeneity at multiple scales (Bar Massada et al. 2008). I will use both medium and fine resolution imagery to calculate two classes of habitat structural measures: a) texture measures derived directly from raw satellite imagery and b) a suite of non-correlated landscape ecology metrics derived from classified (e.g. woody vegetation, grassland, and bare ground) imagery .

The Landsat imagery will be used for the medium-scale analysis. The 30-m pixels are likely too coarse to capture much of the structural differences between the two sites, and fine-scale resolution imagery is probably more appropriate for analyzing structural heterogeneity in this environment. There are several data options for the fine-scale analysis. First, aerial photographs for the sites do exist, but it is unlikely that we will have access to them. A second option is that high-resolution imagery on Google Earth will be made available during the next year. Third, the new THEOS satellite imagery includes a panchromatic band of 2-m resolution. THEOS started operation in October 2008 and products will be available in April 2009.

A fourth option is to take digital photographs of the landscape, delineate vegetation patches, and calculate landscape ecology metrics (Rhemtulla et al. 2002; Bar Massada et al. 2008). I may be able to employ more sophisticated analysis using a new software, Photosynth. Photosynth can merge digital photos taken from various angles into a multi-dimensional mosaic (www.photosynth.com). Currently, methods to extract 3-d and topographic information from the data clouds generated during the "synthing" process are being developed (see http://getsatisfaction.com/livelabs/topics/pointcloud_exporter; http://www.evolutionbeach.org/2008/11/photosynth-point-clouds-topography.html; http://binarymillenium.com/2008/08/exporting-point-clouds-from-photosynth.html).

Chapter 2: Expected Outcomes and Significance

This research will produce these main findings:

- Extent and pattern of shrub encroachment
- Results of the chronosequence analysis to 1) describe post-burn succession of alpine shrub-meadow mosaic and 2) plant biodiversity trends in post-burn succession and for burned versus unburned plots.
- A comparison of the spatial heterogeneity between sites AP1 and AP2.

Findings of this chapter will be summarized in two manuscripts. One paper will be submitted to Remote Sensing of Environment and include a comparison of the classification-based versus the SMA approaches to quantify shrub encroachment. It will include a description of the methods, results, and accuracy assessment. It will also summarize the patterns and drivers of the observed change, and discuss the use of gradient analysis to discern climate-induced change. The other manuscript will be submitted to Landscape Ecology. It will summarize the results at all spatial scales. It will compare whether the burn ban influences spatial heterogeneity at the patch, landscape, and regional scales, and discuss the importance of studying heterogeneity at multiple scales.

Human land use is typically assumed to be harmful for biodiversity, but this research addresses the possibility of a land use that may enhance biodiversity. Thus, this research has significance for environmental policy in southwest China and world-wide, because it explores the actual effects of policy meant to protect the environment. In addition, it addresses policy implications by quantifying field and remotely-sensed measures of change at multiple spatial scales. Not only does this better able to detect on-the-ground change and policy implications, it also helps us to better understand the interaction between and appropriateness of different scales of measurement and analysis. Finally, the research investigates the threat of climate change in sensitive mountain environments, which can be used to inform science and policy in other high-elevation regions around the world.

CHAPTER 3

Research question:

Have China's forestry policies successfully protected high-diversity forests in NW Yunnan?

The natural temperate forests of NW Yunnan are some of the most diverse in the world, and highly threatened. The prominent natural forest types are montane conifer and mixed forests, containing fir, spruce, pine, larch, evergreen oak, birch and rhododendron (Li and Walker 1986). Over 7,000 species of vascular plants have been documented in NW Yunnan, many of which are endemic to the native forest ecosystems (Chang-Le et al. 2007). In addition, natural forests are home to several rare species of pheasants and a viable population of the endangered Yunnan snub-nosed monkey (Xu and Wilkes 2004).

Currently, forests cover over 60% of NW Yunnan (Weyerhauser et al. 2005), but high-diversity natural forest ecosystems are a small proportion (personal observation). The majority of the forests consists of homogeneous stands of early successional pine, oak shrub, and white birch forests. The condition of NW Yunnan forests can be traced to a turbulent history of forest policy and land tenure since the Mao era. From the 1950s through the 1990s, southwest China's forests were used as a key resource in China's development, and forests were clear-cut by state logging companies (Harkness 1998; Morell 2008). NW Yunnan's forests were some of the last to be exploited, with the most widespread logging occurring after 1980 (Zackey 2007), field conversations). State logging was a key part of the economy in NW Yunnan for several years, providing stable employment for local villagers, but decimating old-growth forests (Melick et al. 2007).

After a disastrous flood of the Yangtze River in 1998, the National Forest Protection Plan (NFPP) was implemented. The NFPP imposed a ban prohibiting logging of natural forests in SW China, and allots US $15 billion over 10 years for forest protection and reforestation programs (Weyerhauser et al. 2005; Wang et al. 2007). Statistics from the Chinese government tout the logging ban as a success, and claim broad-scale afforestation in southern China since 1998 (Wang et al. 2007; Morell 2008). Indeed, the logging ban has successfully prevented state-sponsored logging (Harkness 1998). However, 74% of forests in southern China are owned and managed by village collectives (Harkness 1998; Weyerhauser et al. 2006), and studies suggest that the logging ban has not effectively protected these forests. Case studies indicate that logging in remote forests by local people has increased since the logging ban (Weyerhauser et al. 2005; Weyerhauser et al. 2006; Melick et al. 2007; Zackey 2007), and shrubs and secondary growth that could replace these forests are heavily used by local villagers, hampering forest regeneration (Willson 2006; Trac et al. 2007).

My field observations corroborate the story told in the literature. On the ground, the massive deforestation from the state-logging era is obvious because much of the logging took place along roads. Eroded hillsides with grass and shrub vegetation intermixed with large stumps of spruce, fir, larch and oak forest are a common result of clear-cutting by the state logging companies. I observed no recent road-side logging activity, but visited several remote sites where local people were engaged in small-scale logging activities that were gradually clearing mature forests. In general, the new forests have low plant species and structural heterogeneity, and harbor little of the animal biodiversity for which the region's natural forests are known. Soils in the successional forests are often compacted and eroded, and the forests themselves are heavily utilized for subsistence activities by local people, especially livestock pasture and firewood collection. Plant diversity is likely limited by livestock's preferential grazing, firewood harvest, and soil compaction, and these new forests may have limited potential to regenerate into high-diversity forest ecosystems if current use levels continue.

In summary, case studies from the literature and my own observations suggest that the high rates of afforestation reported by Chinese statistics do not represent forest ecosystem dynamics in a way that is meaningful for biodiversity conservation. Although forest cover may indeed be increasing, *I hypothesize that the forest cover increase represents an increase in secondary forest with limited biodiversity value, while the remaining mature, natural forest with high biodiversity value continues to disappear (H3).*

Landsat images cannot be used to directly measure plant or animal diversity, structural heterogeneity, or other direct measures of forest ecosystem biodiversity. Furthermore, the accurate mapping of forest succession is very challenging with Landsat imagery, even under the best of circumstances (Song and Woodcock 2003). Therefore, I will focus on using remote sensing to measure stand-replacing disturbance. Logging of mature forests is the primary source of stand-replacing disturbance in this study area. Broad-scale natural disturbance, such as wildfires, are not common in the study area, and although secondary forests are used for firewood, this typically occurs as single-tree harvesting. I will employ two change-detection methods within the forested regions of the study area to measure stand-replacing disturbance as a proxy for logging. I will then compare rates and patterns of logging before and after the logging ban, in order to determine the ability of the ban to prevent the clearing of the mature, natural forests of high biodiversity value for which NW Yunnan was designated a biodiversity hotspot.

3.1 Change Detection – Change Classification

First, I will perform a multi-temporal composite analysis, in which images from different years are put into a single stack, and then classified into change classes. The analysis will result in four change classes: No Change (1990-2009), Forest Loss 1 (1990-2000), Forest Loss 2 (2000-2009), and Regeneration (1990-2009) classes. The results will provide a broad-scale look at logging disturbance before and after the logging ban. For example, I predict that prior to the logging ban in 2000 (1990-2000), forest-clearing disturbance occurred at a high rate and in the most accessible areas (along roads). I predict that after the ban (2000-2009), cutting of natural forest continued at a much lower rate than previous time periods, and in more remote and less visible areas.

The disadvantage of the multi-temporal composite analysis is that the temporal resolution of such an analysis has to be relatively coarse (e.g., 5-10 years) because the selection of adequate numbers of training sites is not feasible for image datasets with higher temporal resolution. However, disturbance signals are likely to degrade during the relatively long time lag between the images (Healey et al. 2005). In my study area the 10-year time lag (i.e. 1990 – 2000 – 2009) will likely introduce error into the analysis. For example, consider two different forests, one that was clear-cut in 1991 and the other clear cut in 1999. The disturbance signal from the image analysis will be much stronger for the area logged in 1999, even though the actual disturbances were identical. In essence, the coarse temporal scale of the analysis will limit my ability to accurately identify where and when forest clearing occurred.

3.2 Change Detection: Identification of Disturbance using Trajectory Analysis

Recently, NASA has released their Landsat archives to the public, which greatly increases image availability and thus the potential for finer-scale temporal analyses. The development of methods that can take advantage of dense records of Landsat imagery is an important research question for the remote sensing community. Simply adapting the multi-temporal composite analysis technique may not be appropriate, for at least two reasons. First, the method requires training and reference data for each change class. It is usually impossible to get such field-based data, and would be very time-consuming to obtain image-based data for a dense time series. Second, the classification-based analysis relies on changes in the near and mid-infrared image bands from year to year. These same bands are very sensitive to phenological change, requiring the use of anniversary-date images, which may not be available.

Recently, a new technique has been developed to take advantage of dense records of imagery (Kennedy et al. 2007). A single band is used to represent forest cover (e.g. Band 5), and the dense time-stack of images provides a trajectory of spectral reflectance over time for a given forest pixel. The plot's trajectory is then compared to a small number of hypothesized trajectory "models" such as a) disturbance without regeneration, b) disturbance with regeneration, or c) revegetation to a stable state. Thereby the disturbance history can be determined for all pixels whose trajectory statistically matches one of the models. This technique is very exciting because it can characterize change at a much finer temporal scale (e.g. 1 year), allowing a much richer analysis of forest disturbance and regeneration dynamics.

There are at least two obstacles for the use of trajectory based disturbance mapping in my research. First, a *very* dense (preferably annual) stack of images is required, in order to construct and test trajectories of disturbance and revegetation (Kennedy et al. 2007). Such an image dataset is simply does not exist for my study area. Second, since the trajectory analysis is based on the year-to-year change in Band 5 reflectance, anniversary date images are essential where forests undergo seasonal changes. Even with the newly-opened Landsat archives, such dense records of anniversary date images do not exist for NW Yunnan.

Thus, although Kennedy et al.'s (2007) work is exciting, there remains a need to develop methods that take advantage of dense series of *non-anniversary date* images to detect fine-scale temporal patterns of forest disturbance. A promising approach is a method developed by Dr. Phil Townsend and his former student, Clay Baros. The method mitigates seasonal variability by comparing reflectance values of a forest plot at a given date to the average reflectance value of other same-class pixels on the same date. Each pixel is normalized according to its deviation from the average reflectance of like pixels, so anniversary dates are not necessary. A pixel that has an unusually high or low reflectance compared to the average indicates a disturbance. Incorporating each normalized image into a temporal record identifies years of disturbance, and determines a pixel's propensity towards change or stability.

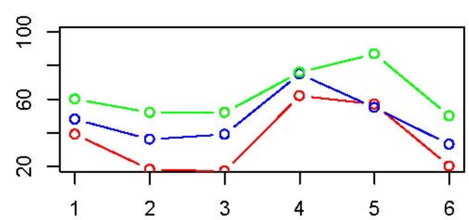
Table 4. Images to be used for the trajectory analysis.

Year	Month
1989	Jan/Feb
1990	November
1991	April
1993/94	Nov/Dec
1996	December
1999	October
2000	April
2000	December
2002	November
2003	February
2004	November
2005	November
2009	Spring
2009	Spring

The normalized image trajectory removes seasonal variability. Other sources of variability that drive forest change, such as climatic variability and topographic position, can then be removed by using canonical correlation analysis (CCA). The statistical relationship between the image reflectance (dependent variables), and topography and climate (independent variables) is quantified in order to determine how much of forest plot's change is driven by "natural" forces of topography and climate. The results of the normalized change trajectory analysis and the Canonical Correlation Analysis are then synthesized, and used to identify change that is not attributed to climatic, seasonal, or topographic variation. This method was used successfully to identify deciduous forest disturbance using 14 non-anniversary date Landsat images in the Appalachian mountains of Maryland (Baros and Townsend 2004).

For my study area, free images from NASA's archive, together with images purchased from the Thailand and Chinese ground stations, consist of 14 images from fall, winter and spring for the 20-year time period between 1989 and 2009 (Table 4). Spectral plots of the multi-temporal imagery demonstrate that there are phenological differences in forests between fall, spring and winter (Figure 7). The use of Kennedy et al's (2007) trajectory-based technique is not appropriate because of the seasonal variation. Therefore, I will use the alternative method that was effective to identify forest disturbance using non-anniversary date imagery in the Appalachians (Baros 2004). A finer temporal scale analysis would not only improve my ability to identify disturbance, but also greatly increase the depth and breadth of my understanding of how the logging ban was implemented, regulated, and adhered to. For example, I will ask such questions as: *did accelerated rates of clear-cutting occur just prior to the burn ban, in anticipation of an impending restriction in logging rights? Was the logging ban effective immediately, or did it take a few years to regulate?* Answers to these questions will highlight people's responses to regulation and the best ways to enforce such regulations.

Figure 7. Spectral profile of a forest pixel in three different seasons of 1999-2000, with the raw image DN on the y-axis and the spectral band on the x-axis. Red – October, Blue – December, Green – April. The reflectance of Bands 4 and 5 relative to other bands are variable, indicating that Bands 4 and 5 are sensitive to seasonality.



Technical Approach

3.1 Change Classification

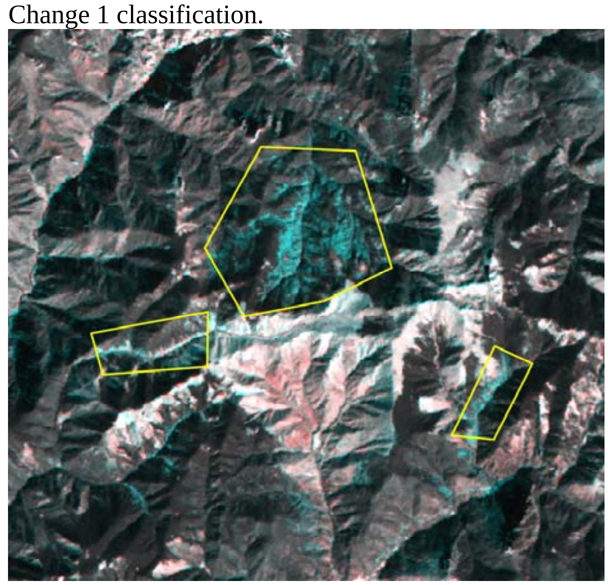
I will perform a change classification in a multi-temporal composite analysis over 3 time periods for which I have anniversary-date (autumn) images. The 3-date imagery will be classified into No Change, Forest Loss 1 (1990-2000), Forest Loss 2 (2000-2009), and Regeneration (1990-2009).

Image pre-processing will follow the procedures described in Chapter 1. I will mask forest/non-forest based on the 1990 classification performed in Chapter 1. In order to include forests disturbed prior to 1990, I will look for permanent clearings versus temporary disturbances by comparing the forest/non-forest map from 1975 with the forest/non-forest map from 1990. I will include areas that were deforested between 1975 and 1990 in the change analysis.

Structures derived from tasseled cap transformation, including brightness, greenness, wetness and the Disturbance Index (DI), will be used in the classification. Tasseled cap structures are especially effective to detect stand-replacing forest disturbances (Healey et al. 2005), and partial harvests (Healey et al. 2006). The relative performance of these depends on forest recovery rates and sampling intervals

(Healey et al. 2005). The Disturbance Index works best when the sampling interval matches the forest recovery rates, and the disturbance signal has not decayed. More complex structures, such as Brightness + Greenness, perform better when forest recovery is rapid relative to the sampling interval, because multi-band structures are better at characterizing the variability of regenerating forests. At this time, I am unable to predict whether the 10-year sampling interval of my imagery matches forest recovery rates for my region. Therefore, I will perform the classification analysis with both the DI and the Brightness/Greenness stacks. DI and Brightness/Greenness will be derived from fall images in 1990, 2000, and 2009. I will stack DI from each year into a 3-layer stack. I will stack the brightness and greenness of each year into a 6-layer stack. Both of these stacks will be classified using the same methods and training data.

Figure 8. Forest disturbance between 1990 and 2000 is highlighted by displaying a layer stack of 1990 Band 5, 2000 Band 5, and 2000 Band 5 as a RGB image. In this image, red areas indicate vegetation regeneration, gray areas indicate no change, and blue areas indicate disturbance. The yellow polygons draw attention to 3 general areas where polygons of bright blue pixels will be chosen for training and reference data for the Change 1 classification.



I will use a combination of field-based and image-based data for training and accuracy assessment. For the No Change class, I can use field data of areas of mature trees. Furthermore, for Forest Loss 2 (Deforestation 2000-2009), I will use field data collected for areas that were obviously recently deforested (in the past 0 to 3 years). All other training and reference data will be derived from the images themselves (Healey et al. 2005; Kuemmerle et al. 2007). Fortunately, spectral signals of deforestation are strong and easily visible (Figure 8). I will select polygons of image-based change classes from the imagery, for use in training and accuracy assessment. I will produce an error matrix for all classes.

3.2 Trajectory Analysis

I will apply the experimental trajectory analysis method used to detect forest disturbance in the in the Appalachian Mountains (Baros and Townsend 2004). Image processing will be performed as described in Chapter 1 for the entire series of images (Table 4). I will produce a single forest/non-forest mask as described for the Change Classification, and apply this mask to all images in the series. Tasseled cap brightness, greenness and wetness will be calculated for each image. The trajectory analysis will then be performed as follows:

Part 1: Create deviation maps that remove seasonality differences:

1. The derived images will be stratified into 4 forest classes based on the 1990 classification: pine, spruce/fir, evergreen oak, and deciduous.
2. The same training areas used for each of the four forest classes during image classification (Chapter 1) will be identified on each image in the series. I will extract reflectance (brightness, greenness and wetness) data for these training area locations (plots).
3. The reflectance values for each plot within a scene will be subtracted from the average reflectance value of all the plots in the same forest class (figure 9a).The standard deviation of reflectance values is the unit of measurement.

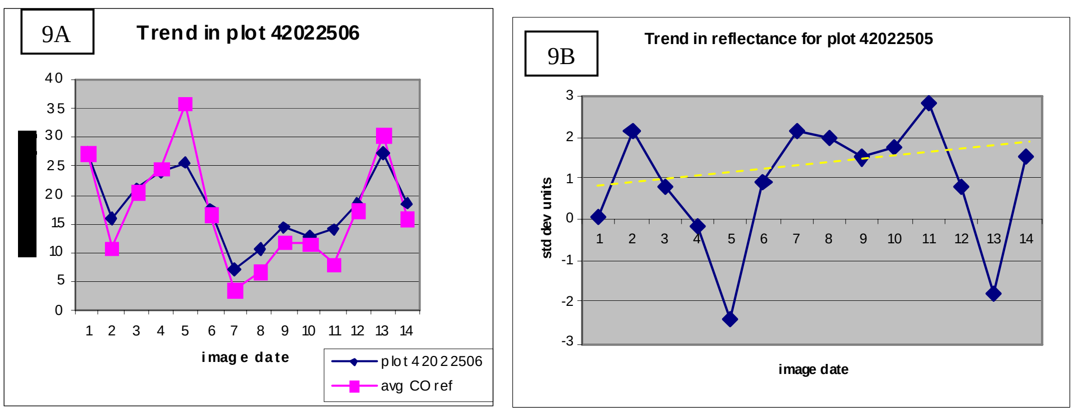


Figure 9. (a) The comparison of one plot with all other plots like it (chestnut oak in this example) provides the deviation of a location from normal conditions on a specific date (y-axis represents percent reflectance). (b) The magnitude of those deviations over time can reveal a trend away from the average conditions and highlight locations of potential forest change (or alternatively, forest stability). Here, the trend in near infrared reflectance is normalized as the standard deviation from mean reflectance. The normalization is shown in 3B where zero standard deviation units indicates little difference from the mean. This example shows a large potential change happening between dates 5-7 and 11-13 and a relatively stable period from date 7-11 (from Baros 2004).

4. For each plot, I will calculate a measure of the plot's overall trend of change over time (trajectory index, yellow line in Figure 9b) and its tendency to deviate from the average change throughout the trajectory (sinuosity index, yellow line divided by blue line in Figure 9b).
5. I will plot the sinuosity index and the trajectory index for each plot to investigate the plot's trends of change (Figure 10).
6. I will create sinuosity index and trajectory index maps for all pixels in the image extent. These maps are referred to as the deviation maps.

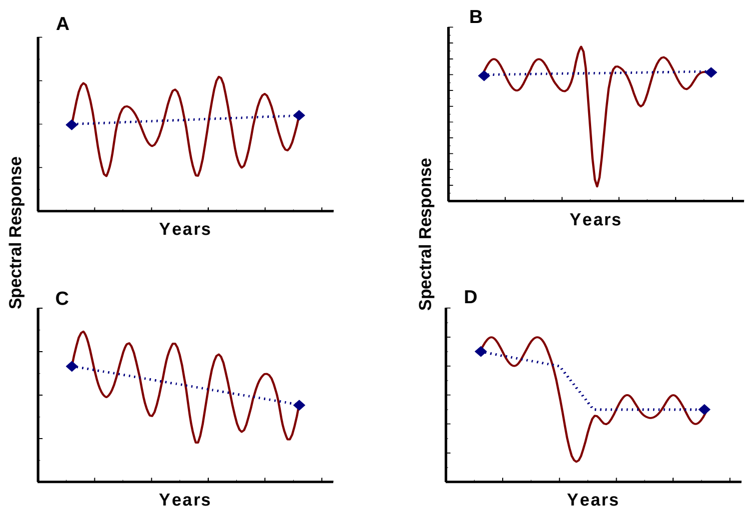


Figure 10. Trend lines of spectral response (i.e., greenness, etc.) for Landsat imagery have both a general direction (blue line) in which they are trending and a distinctive pattern (red line) of change going on in the interim (between the endpoints). Thus, a curve's sinuosity and overall change can both represent a plot's propensity towards change depending on the temporal scale of focus. In this example, Graphs A and B represent more stable long-term trends than Graphs C and D, but there appears to be more short-term change happening in Graph B (from Baros 2004).

Part 2: Use Canonical Correspondence Analysis to remove changes caused by climate and topography:

7. The sinuousity and trajectory indices will be calculated for each of the brightness, greenness and wetness bands, for a total of 6 measures for each plot. These six measures will serve as dependent variables (representing forest change) in the canonical correlation analysis.
8. Forest reflectance changes naturally due to climatic variability and microclimatic conditions related to topographic variation. Therefore, I will use a suite of climatic and topographic variables as independent variables in the canonical correlation analysis. Elevation, slope, and aspect will be derived from the SRTM DEM. I will obtain temperature and precipitation variables at a coarse scale (e.g. at least 1 km) and re-sample them to the 30-m resolution of the Landsat images.
9. Canonical correlation will be performed between the dependent variables (6 reflectance-based variables) and the independent (climatic and topographic) variables. The canonical correlation model is built from plot-based training data. Canonical correlation calculates a single canonical variate for each group of dependent and independent variables, and measures the correlation between dependent and independent variates for each plot. The correlation coefficient between the two variates indicates how much of the spatial variation in the reflectance values is related to climatic or topographic variables. The canonical correlation analysis produces standardized coefficients and loadings for each of the dependent and independent variables.

Part 3: Synthesis of Parts 1 (deviation maps) and 2 (canonical correlation analysis)

10. I will apply the results of the canonical correlation model to the entire image extent. Specifically, the six standardized coefficients from each of the dependent variables

(Brightness Trajectory Index, Brightness Sinuousity Index, Greenness Trajectory Index, Greenness Sinuosity Index, Wetness Trajectory Index and Wetness Sinuosity Index) will be used algebraically with each of the six deviate maps to derive six maps of canonical change variates. Each of these six maps demonstrate that variate's propensity towards change that is not attributable to climate or topography. The six maps are then added together to synthesize the measures of change.

11. The resulting map represents a gradient of disturbance that is not attributable to climate or topography. I predict that the highest values on the disturbance gradient are areas of stand-clearing disturbance, i.e. logging. I will identify a logging threshold by using field data of areas with different general logging histories, including a) older (e.g. in the 1980's) clear-cut with no regeneration, b) older clear-cut with pine regeneration, c) recent (e.g. 2008) clear-cut d) recent partial-cut, e) secondary forest > 25 years old, and f) mature forest. I will find these areas in the final change map and derive a threshold, above which indicates logging and below which indicates absence of stand clearing disturbance. I will apply the threshold to the final change image to identify areas of logging throughout the study area.

12. Finally, for the logged areas, I will identify the time period (e.g. between the 2000 and 2002 images) when logging occurred. I will synthesize a) the final change image that identifies logged areas (step 11) with b) the deviation trajectories (step 5, figure 10) to produce a layer indicating the time period of disturbance for each logged pixel.

Part 4: Accuracy Assessment

13. Detailed logging history is not available for the study area. Therefore, I will assess the accuracy of occurrence and date of logging disturbance in three ways. First, I will compare the results of the trajectory analysis with both the post-classification change analysis from Chapter 1 and the multi-temporal composite change classification from Chapter 3. Although these two other analyses are performed at a coarser temporal resolution than the trajectory analysis, general agreement will indicate a rudimentary level of accuracy. Second, in the field I observed several sites in the process of being logged, and will likely visit additional sites in the future. I will determine whether the trajectory analysis accurately identified logging observed in the field. Finally, I will use visual interpretation of the satellite images to perform a quantitative accuracy assessment (Kennedy et al. 2007). For example, I will randomly select pixels from each time period of the analysis. I will assign each pixel as "no-change" or "logged" based on visual interpretation of the tasseled-cap transformed Landsat images. Field and image data used in the accuracy assessment will be independent from the data used to identify thresholds in step 11.

Ultimately, I will compare rates and patterns of disturbance quantified from both the Disturbance Index method and the trajectory-based method. Using a DEM and GIS layers of road networks, villages and town locations, and provincial and township boundaries, I will determine temporal and spatial patterns of change and the drivers of the change. For example, I will test my hypothesis that, overall, deforestation rates decreased following the logging ban. I predict that prior to the logging ban, areas closest to roads had the highest disturbance rates. I also predict that immediately following the logging ban 1) forest clearing along roads decreased, but 2) forest clearing rates in more remote areas increased. I also predict patterns of forest disturbance will vary with topography (slope, elevation, aspect) and political units (township). For example, I predict that the most topographically rugged areas (e.g. high elevation and slope) suffer fewer disturbances. In addition, I predict that the enforcement of the logging ban varies by township, the smallest administrative unit of regulatory agencies.

Chapter 3: Expected Outcomes and Significance

Findings of this chapter will be summarized in a manuscript submitted to Remote Sensing of Environment. It will include a description of the two different methods to identify disturbance, and a discussion of the benefits and disadvantages of each method. It will summarize the patterns and drivers of forest disturbance observed over the study period, and will assess the effectiveness of the logging ban to protect natural forests. This research is relevant to the wider remote sensing audience because it compares two very different methods of forest change detection. One of the methods is new, and if it works, will be an exciting step towards fully exploiting NASA’s recently released Landsat record. The ability to use this free image database has particular relevance for scientists in developing countries who lack resources to purchase imagery. The ability to exploit non-anniversary date imagery is relevant for all scientists, because such imagery is all that is available for many study areas. Finally, the research has significance to LULCC scientists because it explores the implications of national environmental policy in a unique part of the world that has not yet been studied.

CHAPTER 4

Research question:

Have government or cultural forms of protected areas been more effective to protect threatened ecosystems?

Over the past 25 years, protected land has increased exponentially. Worldwide, there are currently more than 100,000 protected areas (i.e. national parks, provincial parks, nature reserves, game reserves) encompassing 11.5% of the earth's land surface (Naughton-Treves et al. 2005). Much of this increase has occurred in developing countries that are high in biodiversity. These same developing countries are also undergoing rapid development and population growth, and often biodiversity is located in the same areas where people live, creating a conflict between the needs of local people and conservation (Zimmerer et al. 2004; DeFries et al. 2007). This conflict often results in minimal protected area "effectiveness", i.e. ability of a protected area to protect its target ecosystem.

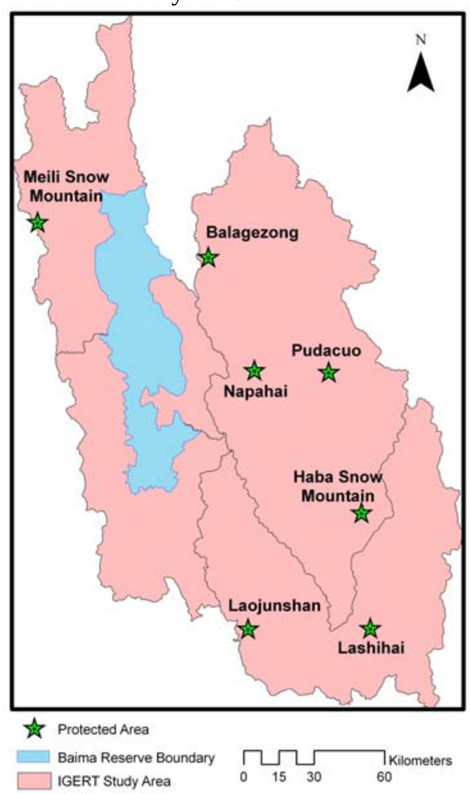
The limited effectiveness of protected areas was highlighted in a 2001 Science paper that showed the failure of China's Wolong Nature Reserve to protect Giant Panda habitat (Liu et al. 2001). Since then, much attention has been focused on assessing protected area effectiveness, worldwide. The establishment of protected areas in many regions can merely create "paper parks", where lack of enforcement and management leads to ineffectual reserves (Joppa et al. 2008). Many tropical parks successfully protect forests within their boundaries (Bruner et al. 2001), but this is often counterbalanced with "leakage", or accelerated degradation outside of protected area boundaries (DeFries et al. 2005; Ewers and Rodrigues 2008; Joppa et al. 2008; Wittemyer et al. 2008). Leakage results in protected area isolation, which disrupts ecological processes essential for long-term ecosystem stability and species' viability (Hansen and DeFries 2007). Despite these sobering results, there is evidence that effectiveness can be improved and leakage be mitigated by designing management systems that balance biodiversity conservation with the needs of local peoples (Nagendra et al. 2004; DeFries et al. 2007).

<u>Protected Areas in China</u>

The issue of protected area effectiveness is especially relevant in China. Since the 1980s, China's protected area system has expanded greatly, adding 1500 nature reserves nationwide while undergoing population growth and massive economic development. China's protected area system adheres to a strict hierarchy. Scenic areas are designated in name only, as they receive no state financial support. County, prefectural and provincial-level (e.g. Yunnan province) reserves are funded and regulated at increasingly higher administrative levels. National-level nature reserves, of which Wolong Giant Panda Nature Reserve is one, top the hierarchy. Finally, new additions to the system are "National Parks", promoted by The Nature Conservancy. These are designed to encourage organized tourism (i.e., with roads, hiking paths, restaurants and lodging) from which local communities can benefit. One national park was created in NW Yunnan in 2006, as a model for all of China. Nationwide, there are already four more parks on the drawing board.

There is wide acknowledgement that protected areas throughout China face serious obstacles (Li and Han 2000; Liu et al. 2003; Tang et al. 2006). Challenges include the inappropriate design of many reserves, a lack of financial support provided for administration, and the conflicts between resource protection and the historical rights of local people to use these resources. Implementation of a nature reserve in China can involve the relocation of local people, and restriction of local communities' historic rights to natural resources. Specific land tenure rights and restrictions are typically obscure and ever-changing (Xu and Melick 2007), and the interactions between nature reserves, wildlife, local people, land tenure rights, and economic development in rural China are complex. As demonstrated by the case of the Wolong Nature Reserve's attempt to protect Giant Panda habitat, failure can occur even when reserves enjoy ample financial and governmental support (Liu et al. 2001).

Figure 11. Location of protected areas in the IGERT study area.



<u>Protected Areas in NW Yunnan</u>

Since 1980, several protected areas have been created in NW Yunnan to protect one of China's biodiversity hotspots, and several new protected areas are pending (Figure 11, Table 5). Baima Nature Reserve is NW Yunnan's largest and oldest reserve, established in 1983 to protect the endangered Yunnan Snub-nosed monkey and its temperate forest habitat. Bitahai was established in 1984 at the prefecture level, and then in 2006 expanded and transformed into Pudacuo, China's first National Park. Pudacuo protects a mosaic of ecosystems including mature and regenerating temperate forest, sub-alpine meadows, and alpine lakes. Other protected areas include Napahai and Lashihai Nature Reserves, created to protect high-elevation wetlands crucial for black-necked cranes and other threatened migratory waterfowl. Laojunshan is a mountain summit in the southern region of the study area that was designated in 1986 at the prefecture level to protect temperate forests and alpine meadows. Laojunshan, Balagezong, and Meili Snow Mountain area all slated to be placed under national protection in the next few years.

All of these protected areas are inhabited by subsistence-based peoples who have had historic rights to the resources for centuries. Many scientists working in NW Yunnan question whether government-sponsored nature reserves are appropriate in a region that is densely populated by indigenous communities who have sustainably managed these ecosystems for centuries (Harkness 1998; Bo et al. 2003; Xu and Wilkes 2004; Yang et al. 2004; Xu et al. 2005; Xu and Melick 2007). The scientists present case studies that demonstrate how national policies can destroy the existing locally-based management systems, without replacing them with effective alternatives. They clearly demonstrate serious local-scale environmental degradation, including illegal logging, overuse of community forests, and overgrazing. However, no attempt to quantify the consequences of NW Yunnan's protected areas at the regional scale has been made to date. A regional-scale assessment of protected areas is critical, in light of the central government's plans for massive investment in the expansion and upgrade of NW Yunnan's protected area system in the next several years.

Table 5. Timeline of protected area implementation in the IGERT study area.

Year	Event
1980s	Meili Snow Mountain designated a scenic area
1983	Baima Nature Reserve established at the provincial level
1984	Bitahai, Napahai, and Haba established as prefecture-level nature reserves
1986	Laojun Mountain designated as prefecture-level nature reserve
1988	Baima Nature Reserve elevated to national-level protected area
1998	Lashihai Nature Reserve established
2000	Baima NR expanded into Weixi County
2003	UNESCO inscribes Three Parallel Rivers of Yunnan World Natural Heritage Site
2006	Pudacuo, which encompasses Bitahai NR, designated a National Park
2007	Balagezong opened for ecotourism
2012	Balagezong will be designated as a National Park
2012	Baima Nature Reserve will be designated as a National Park
Application pending	Laojun Mountain will be designated as a National Park
Application pending	Meili Snow Mountain will be designated as a Provincial Nature Reserve

Protected areas analyzed in this study

I will quantify regional-scale protected area effectiveness in NW Yunnan by focusing my analysis on protected areas that 1) are contained within my Landsat image footprints, 2) have a history of official protection long enough to warrant analysis, and 3) I can obtain reliable protected area boundary maps. Thus far, only Baima Reserve satisfies the final criterion, since the Chinese government is very sensitive about the acquisition of maps and GIS data by foreigners. Baima Reserve boundaries are in hand (Figure 11), and Napahai, Bitahai/Pudacuo and Laojunshan will all be included in the analysis if I can obtain a reasonable estimate of the reserve boundaries.

<u>Baima Reserve</u>

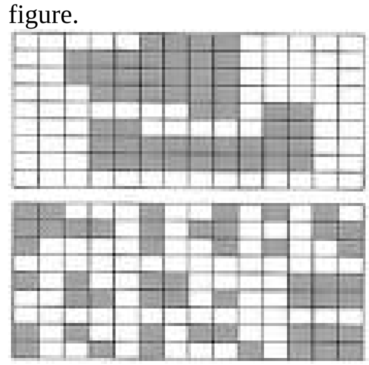
Baima is the region's flagship reserve, implemented to protect the habitat of the endangered Yunnan snub-nosed monkey. Baima is NW Yunnan's largest protected area, encompassing 2800 km^2 (12% of the study area) and contains most of the region's remaining old-growth temperate forests, over half of the world's remaining snub-nosed monkeys (China 2003) and one of the most highly diverse alpine meadow ecosystems in Yunnan (Sherman et al. 2008). Threats to the reserve include logging, hunting, and subsistence land use by local peoples (Xu and Wilkes 2004). Baima was established as a provincial-level nature reserve in 1983 and upgraded to a national-level reserve in 1988. In 2000, the reserve's size increased by approximately 20%, as it expanded south to include several more groups of the snub-nosed monkey. The Baima Reserve is slated to become a National Park in 2012.

Baima and its border region are populated by ethnic minorities (78% of whom are Tibetans) who have lived in the region for centuries and employ traditional subsistence agro-pastoralism. There are 10,000 people in 13 villages within the reserve itself, and 60,000 people in 41 villages within the 3-km border region of the reserve (China 2003). The current rules regarding resource use by local peoples in the Baima Reserve are unclear, but conversations in the field indicated that at the least, logging and burning restrictions are more rigidly enforced within Baima compared to the rest of the landscape. *I predict that within the reserve, deforestation continues, but at a lesser rate than in the rest of the study area (H1).* However, since local people rely on local resources for their livelihoods, the decreased use of reserve resources must be counter-balanced with increased use resources outside of the reserve. Furthermore, some areas within the reserve are more remote than others, and therefore less subject to enforcement. *I hypothesize that the protection of forests within the reserve has resulted in "leakage", i.e., increased degradation in the reserve buffer zone, and in remote, inaccessible regions within the reserve (H2).*

Rate of Change versus Spatial Pattern

In addition to the rates and trajectories of land cover change, the spatial distribution of land covers is important for biodiversity (Figure 12). First, fragmentation breaks large patches of habitat into smaller patches. In general, a larger habitat patch will maintain higher species diversity (MacArthur and Wilson 1967). Second, spatial pattern of a habitat patch, including the amount of edge and interior habitat, have important implications for individual species' viability. For example, some forest bird species are interior specialists, whereas other avian species thrive at the forest edges. Finally, habitat connectivity is important because the ability of organisms and populations to move between patches enhances the probability of long-term species viability. The most obvious form of connectivity is corridors of similar land cover. However, the non-habitat matrix is often a mosaic of various land covers, some of which are impenetrable while others easily facilitate movement and dispersal (Ricketts 2001). Therefore,

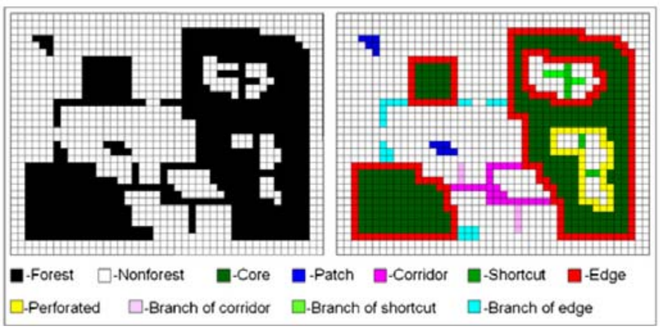
Figure 12. The top and bottom images have the same amount of habitat (gray pixels), but the distribution of the habitat have important implications for biodiversity. Patch size and interior habitat is much greater in the top figure, whereas patches are smaller and more dispersed on the bottom figure.



"connectivity" is not simply a measure of distance between patches, but should incorporate an evaluation of the type and distribution of the matrix.

A common way to synthesize landscape pattern is to classify a landscape into habitat and non-habitat classes, and then generate numerical landscape metrics, such as patch area, patch shape, connectivity, and patch density. The numerical indices can be used in statistical analysis to understand the relationship between landscape pattern and biodiversity. A spatially-explicit method that maps spatial patterns using morphological image processing has recently been developed (Vogt et al. 2007). A binary habitat/non-habitat map is classified into edge, core, perforated, and corridor (Figure 13), based on the spatial context of each pixel. The resulting maps can be used in statistical analysis, and to explore the differences in spatial distribution of habitat at political boundaries or in topographically rugged areas (Kuemmerle et al. 2006).

Figure 13. Morphological image processing is used to classify a binary map into sub-classes based on spatial pattern.



I predict that along with differences in the relative proportion of different land cover types, spatial pattern differs within, at the edges and outside of the Baima Reserve (H3). For example, I predict that the buffer zone of the reserve is the most highly fragmented, with small habitat patches, low amounts of core forest, and high amounts of edge habitat. I also expect that areas within the reserve that were previously logged are regenerating, and thus fragmentation is less within the reserve compared to the rest of the study area. However, I predict that remote areas of the reserve that were previously large native forest patches have received increased pressure from local logging, thereby inducing forest fragmentation, decreased connectivity, and less core habitat.

<u>Sacred Areas</u>

As humans dominate more and more of the Earth's surface, natural habitats diminish in proportion and become fragmented. In response, we have implemented protected areas throughout the world to protect threatened ecosystems. However, as species continue to disappear despite of this ever-expanding network, we are forced to question the protected area paradigm and search for other solutions. Scientists and conservationists have recently begun to look to indigenous cultures and their traditional land management practices for new ideas to maintain high levels of biodiversity in environments used by people (Nabhan et al. 1982; Bennett et al. 2006; Ranganathan et al. 2008). Specifically, several publications have shown that indigenous forms of land protection, such as sacred lands, exist around the world and can effectively protect biodiversity (UNESCO-MAB 2003; Bhagwat and Rutte 2006; Bossart et al. 2006; Nepstad et al. 2006).

Although typically smaller than a government-sponsored protected area, sacred areas in particular hold several advantages over government-implemented reserves. They are community-based and thus compatible with local livelihoods. They are scattered and numerous, and therefore encompass a wide range of habitats and micro-ecosystems. Furthermore, they exist in many regions of high biodiversity around the world (UNESCO-MAB 2003).

SW China, specifically, is a region where sacred areas are potentially an important, if not essential, tool for effective conservation. The region is extremely biologically diverse, and serves as a corridor between the tropics of southeast Asia, the Himalayan mountains, and the northern plains of central Asia. At the same time, SW China is home to several ethnic minority groups that retain well-defined cultural identities. Many of these groups recognize sacred areas as part of their culture and religion. Land use traditions vary between ethnic groups, and even between different villages of the same ethnicity. However, sacred areas tend to have in common that certain activities, such as grazing, cutting, hunting and agriculture, are prohibited. The landscape-scale pattern of remnant, native vegetation patches scattered throughout the region possibly act as habitat oases, dispersal footsteps, and migratory stopovers for a diverse array of organisms. Prominent scientists and conservationists within SW China believe that, in the face of rapid development, it is essential to include sacred areas and sacred knowledge into conservation plans to ensure the long-term viability of the maximum number of species (UNESCO-MAB 2003; Xu et al. 2005; Melick et al. 2007; Xu and Melick 2007).

<u>Tibetan Sacred Areas</u>

Within SW China, NW Yunnan is one of the hotspots of both biological and ethnic diversity. Several of the native ethnic minorities, including Yi, Lisu, Dulong, Bai, Tibetan, Naxi, and Nu ethnicity, use sacred forest patches (Peng et al. 2003), but in general very little is known about the management, distribution, or meaning of these sacred sites. Of all the ethnic groups, the religious beliefs and sacred sites of Tibetan Buddhists are the most studied.

Tibetan spirituality maintains that life is a non-material cycle with no beginning and no end. The belief in reincarnation naturally leads Tibetans to consider trees, flowers, and animals as kin. This is markedly different from the way many cultures view nature and their environment, and is the basis of many traditional Tibetan land use practices (Xu et al. 2005). For example, in the same way that agricultural land, pasture, or community forests have a specific use and designation in Tibetan culture, "sacred land" is also an important land use class. Mountain peaks are sacred to Tibetans, and protected from overuse. Sacred mountains are few in number but large in size (10s to 100s of km^2). They are recognized as sacred by Tibetans and Buddhists throughout Asia, and are destinations for local and regional religious travelers. There are up to six sacred mountains within my study area.

Sacred mountains form the focal points of a complex sacred site system that also include hundreds of community-based sacred forests that are smaller (1 to 1,000 ha) and at lower elevations than the sacred mountains (Peng 2003). Most of NW Yunnan forests were clear-cut during the Mao era. Some areas were reforested with pine plantations, and in others natural regeneration was impaired by erosion and grazing (Xu et al. 2005). Near roads and villages, the sacred forests are the only remaining oases of natural forest (personal observation). Management and religious significance of the local sacred sites are different for each community, but typically religious rituals are carried out at these sites, and certain activities, such as grazing, cutting, hunting and agriculture, are prohibited (Peng 2003). A vegetation survey at several sacred forest sites indicate that they have higher species richness, diversity, and endemism than randomly selected non-sacred sites (Anderson et al. 2005), and trees are larger and denser (Salick et al. 2007). My field observations from visits to thirteen sacred forests corroborate and add to the information about sacred forests from the literature. *I hypothesize that the designation as a sacred forest has protected small remnants of natural forest from the state-sponsored logging era, as well as from subsistence based grazing and firewood collection (H4a).*

Importantly, the delineation of sacred areas is unlike that of state-sponsored protected areas. A protected area has boundaries within which certain activities are prohibited. A sacred area can best be perceived as a focal point with a sphere of diminishing influence. The focal point tends to be a place of beauty (e.g. a towering rocky outcrop, a natural spring, or a summit). If accessible, the focal point is a site of religious activity, with a small altar, prayer flags, and incense. If the focal point is a forested site, the area around the point is considered a sacred forest. Mature forest is characteristic at and near the focal point. In general, as one moves further away from the focal point, one encounters more evidence of human alteration, in terms of cut trees, secondary vegetation, and cattle grazing. However, alteration is also clearly driven by accessibility, so a deep ravine at the edge of the sacred forest could have undisturbed vegetation. *I hypothesize that sacred forests experience a gradient of human disturbance that roughly corresponds with distance from the focal point (H4b).*

Tibetan Land Use Practices – Evolution and Change

It is important to emphasize that although Tibetans employ “traditional” land use practices, these practices are adaptive and continuously evolving. Different villages’ livelihood strategies developed not only according to custom or convention, but also to environmental and socio-economic circumstances. For example, land use practices and landscape pattern vary significantly both due to elevation and road proximity, and livelihood strategies have changed over time as villagers adapt to political changes and infrastructure development over the past 100 years (Salick et al. 2005). As discussed in the previous chapters, recent articles demonstrate that an especially rapid rate of change in land use practices has occurred in the past three decades due to recent governmental policies (Harkness 1998; Xu et al. 2005; Ediger and Huafang 2006; Weyerhauser et al. 2006; Willson 2006; Melick et al. 2007; Xu and Melick 2007). It is unclear whether the land use practices involving sacred areas have been included in this change. *I hypothesize that recent rapid evolution of livelihood and land use practices in the Tibetan landscape has manifested in changes in the meaning, use and composition of sacred forests (H4c).*

Technical Approach

I will use the results of my remote sensing analysis of Chapters 1 and 3 to 1) determine the effectiveness of Baima Reserve to protect forest cover at multiple scales, and 2) measure whether sacred forests are effective to protect forest ecosystems compared to non-sacred forests, and if this effectiveness has changed over time.

4.1 LULCC trends within the reserve compared to the rest of the study area

I will use the results of my LULCC analysis to determine whether rates and trajectories of LULCC were different within the reserve compared to outside the reserve. The reserve was established at the provincial level in 1983, and upgraded to the national level in 1988. I predict that during the first time period of the satellite image analysis (1975 to 1990), LULCC trends within the reserve were similar to the trends for the entire study area (Chapter 1, Table 1). Following 1990, trends within the reserve differed from those in the rest of the study area because of restrictions within the nature reserves. For example, I predict less deforestation and urbanization, and more alpine shrub encroachment and agricultural abandonment, within the Baima Reserve compared to the rest of the study area.

4.2 Forest cover and Reserve Isolation

Next, I will compare forest change within the reserve with forest change in a 10-km buffer zone around the reserve (Bruner et al. 2001; Mas 2005) to address issues of leakage and isolation. I will measure forest change in two ways. First, I will compare deforestation and afforestation rates from Chapter 1 and second, I will compare logging rates from Chapter 3. I predict that forests within the reserve have been well protected since 1990, but that forests at the edge of the reserve have been heavily impacted because local people require wood for fuel and construction. Furthermore, I predict that there are different levels of deforestation within the reserve. My observations in non-reserve areas indicate that, prior to the logging ban, state-sponsored companies clear cut large stands, and preferred areas near roads due to easy accessibility. However, I observed current (i.e. illegal) logging in the field, and this was performed primarily by local villagers in road-less areas using rudimentary equipment (e.g. hand chainsaw and yak-drawn sleds).

I will use shape files of road networks and villages to delineate various zones within the reserve. I will then determine whether rates and patterns of logging in the different zones changed after the logging ban. For example, I predict that forest clearing patterns followed road networks before the logging ban, and that remaining road-side forests have remained since the logging ban. However, I predict that small-scale logging by villagers in zones that are road-less, but relatively close to a village, has increased since the logging ban. I predict that forest clearing in all time periods has been the least in the most remote zones, i.e., those that are very far from both roads and villages.

4.3 Forest cover and Fragmentation

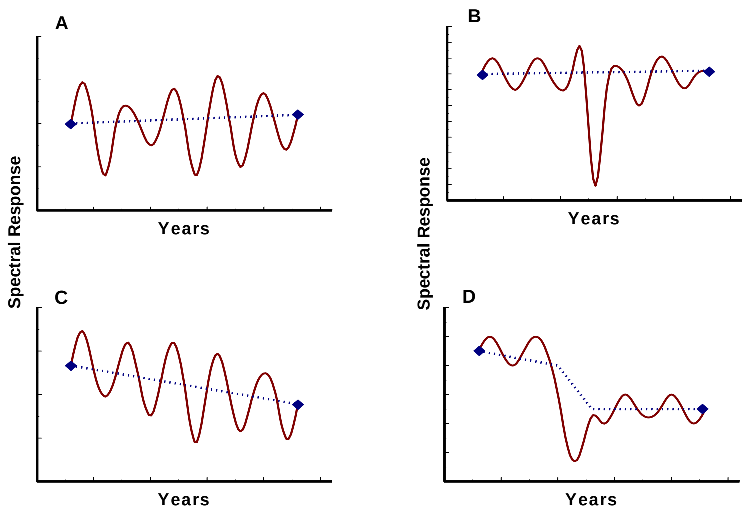
I will also look at changes in spatial patterns of forests. I will assess habitat fragmentation using image morphological processing (Vogt 2006, 2007), which allows the quantification of such basic measures such as patch size and connectivity, and also highlights areas of edge and core habitat, corridors between habitat patches, and perforations within habitat patches. I predict that forest fragmentation has increased in the buffer zone and in road-less zones near villages, whereas fragmentation has decreased in areas of the reserve that were cut prior to the logging ban and are now regenerating.

4.4 Change within and outside of sacred areas

I hypothesize that sacred forests have been more resilient to land cover change compared to adjacent non-sacred areas. I will visit 10 sacred forests within the study area, determine the location of each forest's focal point, and estimate its boundaries based on conversations with local people and visual cues such as vegetation type, vegetation density, and signs of disturbance (e.g. grazing and cutting). I will use the results of the LULCC change analysis from Chapter 1 to determine whether LULCC patterns differ depending on a point's location within or outside of a sacred forest. For example, I predict that points within a sacred forest have undergone less class conversion than points outside of the sacred forest.

I also hypothesize that, within a sacred forest, change patterns will differ depending on the distance from the focal point. A sacred forest has a diminishing sphere of influence from a focal point,

Figure 14. Spectral response from the trajectory analysis in Chapter 3 can be used to formulate hypotheses of forest change within and outside of sacred forests. For example, Figure A represents a pixel near the center of the sacred forest, where both the general direction of change (blue line, Trajectory Index) and the annual variation in change (red line, Sinuousity Index) are relatively stable. Figures B and C indicates two different areas inside and near the edge of a sacred forest. In Figure B an area was cut but then regenerated undisturbed. In Figure C, the forest is thinning out gradually over time due to a gradual increase in the use of sacred forests for grazing and firewood collection. Figure D is an area outside the sacred forest that was logged but did not experience undisturbed regeneration.



and such subtle change will probably be difficult to characterize using categorical variables. Therefore, I will compare change in terms of the continuous variables generated from the trajectory analysis in Chapter 3 (Figure 14). Specifically, I hypothesize that the sphere of influence of sacred forests has diminished over time as national policies and economic development cause shifting livelihood strategies. I predict that near a sacred forest's focal point, both the trajectory and sinuosity indices will be stable because such areas experience very little human-induced change over time. I also hypothesize that the use and meaning of sacred areas have changed over time, and I predict that pixels at the edge of a sacred forest have a higher propensity towards change compared to pixels close to the center because the sphere of influence of the sacred site is gradually shrinking.

Chapter 4: Expected Outcomes and Significance

Findings of this chapter will be summarized in a manuscript targeted for Conservation Biology. It will discuss the ability of the Baima reserve to protect forest and alpine ecosystems, using the results of the different types of image analysis as well as the results of the landscape pattern analysis. It will address issues of leakage and isolation by comparing the reserve with its buffer zone. The manuscript will also address the ability of sacred areas to protect natural forests, and whether sacred forests have remained constant over the time period of the study.

In general, the research has important implications for conservation biologists world-wide, because it explores the dynamics of two very different types of protected areas in a unique part of the world that has not yet been studied. It contributes to the current literature of protected area effectiveness, but adds novel image and spatial pattern analysis techniques in the assessment. The research adds to our very limited knowledge about the meaning, use, and distribution of sacred areas, and it uses remotely sensed imagery as a tool to assess patterns and change within sacred forests, which has not yet been done. From a management perspective, this research has special relevance for conservationists, and policy-makers in SW China, because it assesses protected area effectiveness in a region where considerable expansion of the protected area system is planned.

CHAPTER 5

Research Question:

Question 5: Do traditional Tibetan sacred forest enhance avian biodiversity?

NW Yunnan is a center of avian endemism in China (Lei et al. 2003), inhabited by many species of high-altitude Himalayan birds (personal observation), and an important migration flyway between northern and southern Asia (Wang et al. 2000). Despite its potential importance for avian biodiversity, no research has been done on forest songbird distribution, diversity, migration, population trends, or habitat selection in NW Yunnan. Time and money limitations prevent me from performing an extensive bird survey throughout the study area. I propose to perform a focused study on forest songbird communities in sacred forest patches and the surrounding secondary forest matrix in Shangri-la County.

This research has three primary objectives. First, I address basic questions about avian communities in NW Yunnan. For example, what is the composition of the bird communities that inhabit NW Yunnan forests? What are basic patterns of forest songbird abundance and diversity? What types of forests have highest abundance and diversity, and which forests harbor a reduced avifauna? What are factors influencing forest bird distribution, diversity, and abundance?

The second objective of my research is to assess the contribution of sacred forests to maintain biodiversity at the landscape scale. Although sacred forests are a very small proportion of the landscape (<1%), and surrounded by degraded and homogeneous secondary vegetation, forest bird diversity appears to be high within the patches. Do sacred forests act as “keystone structures” (Tews et al. 2004) that serve to maintain high forest songbird biodiversity in a highly impacted landscape?

Finally, I address whether the "island" or "mosaic" conceptual framework is more appropriate to study forest songbirds in NW Yunnan. For example, island biogeography theory (MacArthur and Wilson 1967) would predict that patch size and isolation are primary drivers of avian diversity in sacred forest patches. On the other hand, traditional measures of fragmentation are not reliable predictors of animal diversity in many terrestrial ecosystems, due to interactions between the patch and matrix (Prugh et al. 2008). In my study area, does the composition and quality of the matrix matter to bird communities?

<u>Sacred Areas</u>

Sacred areas are potentially an important, if not essential tool for effective conservation in SW China (UNESCO-MAB 2003; Xu et al. 2005; Melick et al. 2007; Xu and Melick 2007). The region is quite topographically and biologically diverse, and serves as a corridor between the tropics of southeast Asia, the Himalayan mountains, and the northern plains of central Asia. SW China is home to several ethnic minority groups that retain well-defined cultural identities that recognize sacred areas as part of their religion. The landscape-scale pattern of remnant, native vegetation patches may be a key habitat network that serves to maintain high levels of biodiversity throughout the region.

Within SW China, NW Yunnan is one of the hotspots of both biological and ethnic diversity. Dozens of ethnic groups inhabit the region, and most of these have sacred sites imbedded in their landscape. The sacred sites of Tibetan Buddhists are the most studied. Tibetan sacred mountains are few in number but large in size (tens to hundreds of square kilometers), and form the focal points of a complex sacred site system that also include hundreds of community-based sacred forests that are smaller (1 to 1,000 ha) and at lower elevations (Peng 2003). Management of sacred forests varies by community, but typically activities such as grazing, cutting, hunting, and agriculture are prohibited (Peng 2003). A vegetation survey at several Tibetan sacred forest sites in NW Yunnan indicate that they have higher species richness, diversity, and endemism than randomly selected non-sacred sites (Anderson et al. 2005), and trees are larger and denser (Salick et al. 2007).

Within NW Yunnan, Shangri-la County, and especially its capital city, Zhongdian, is the most economically developed and rapidly growing region. Shangri-la County is historically populated by Tibetan minorities who have relied on subsistence-based agriculture, yak herding and forestry practices for centuries. But as the Chinese central government finances tourism development and incentivizes immigration of Han Chinese from other parts of China, Shangri-la County will continue to grow and change.

Massive clear-cuts of natural forests occurred in Shangri-la County during the Mao era, and natural forest regeneration has been hampered by erosion and Tibetan livelihood activities (Willson 2006). The majority of the landscape is covered by a homogeneous secondary forest of pine and oak shrub. Near roads and villages, the sacred forests are the only remaining oases of natural forest (personal observation). The designation as a sacred forest seems to have protected small remnants of natural forest from the state-sponsored logging era, as well as from subsistence-based grazing and firewood collection.

My field observations in thirteen sacred forests in Shangri-la County indicate that sacred forests vary widely in terms of elevation, size, topography, vegetation, and management. It is clear that the rules regarding the use of sacred forests are not consistent from site to site, but in all cases sacred forests were less impacted by livelihood activities compared to their surroundings. Sacred forests contain a mix of mature and successional vegetation, and its distribution is highly driven by a human disturbance gradient and topographic variation. In general, sacred forests contain higher proportions of tree species representative of the natural temperate forest ecosystem of the region. It is also evident that they have higher overall vegetation species richness and structural heterogeneity compared to the surrounding heavily-used environment. In spite of their miniscule proportion of the landscape (<1%), a high diversity of forest songbirds is present in sacred forest patches (personal observation).

<u>5.1 Habitat heterogeneity and factors driving forest songbird distribution</u>

Habitat heterogeneity in terrestrial systems, i.e., the vertical and horizontal structure and composition of vegetation, is essential to maintain biodiversity because structure provides different niches for animal species at local and regional scales (Belsky and Canham 1994; Tews et al. 2004; Bennett et al. 2006). However, increasing heterogeneity eventually results in habitat fragmentation, which has negative implications for species abundance or diversity. The point at which heterogeneity becomes fragmentation is a critical threshold that is of much interest to conservation biologists. However, it is very difficult to measure because it differs between species, and depends on the scale of measurement.

Worldwide, forest clearing and fragmentation is one of the greatest threats to forest-dwelling animal species, including forest birds. Avian species show a wide range of responses to forest fragmentation, since different species have different habitat needs. For example, edge specialists may thrive in a highly fragmented environment, whereas interior species may disappear (Daily et al. 2001). However, there are general trends in how community-based measures, such as bird species richness and abundance, react to differing levels of forest disturbance and habitat heterogeneity. Specifically, across a variety of forest ecosystems all over the world, plot-scale measures of vegetation heterogeneity, including vertical structure, horizontal structure, and species composition, are consistently the primary drivers of forest bird distribution (Opdam et al. 1985; Estades and Temple 1999; Bhagwat and Rutte 2006; Ding et al. 2008; Ranganathan et al. 2008).

The relationship between plot-scale heterogeneity and bird diversity and abundance is not necessarily linear, as demonstrated by research on bird communities at different successional stages after disturbance. For example, in Taiwan's mountain forests, natural disturbance is historically absent. Forest bird species richness increases along the post-fire successional trajectory and reaches its highest levels in mature forests, probably because local avifauna are adapted to mature forest ecosystems (Ding et al. 2008). On the other hand, boreal forests of western North America are adapted to natural wildfire. Thus, avian diversity and abundance patterns along the successional trajectory are different from those in Taiwan, probably because the avifauna have evolved to take advantage of the different niches provided by post-disturbance successional stages (Schieck and Song 2006). And in eastern North America, the highest avifaunal richness and density occurs in the early successional stages following clear-cutting (Keller et al. 2003). Thus, specific relationships between bird communities and habitat heterogeneity vary widely, and depend on the forest ecosystem and its disturbance regime.

NW Yunnan forests, like Taiwan's mountains, lack a major natural disturbance that has shaped the avifauna of the region. However, local Tibetans have utilized NW Yunnan's forests for hundreds, if not thousands, of years. One would thus expect an avifaunal community that was adapted to both mature and moderately disturbed forests. Fortunately, sacred forests have a great deal of between and within-forest variability in terms of vegetation type and disturbance intensity. Thus, studying vegetation characteristics and bird communities within just a few sacred forests will provide a relatively wide gradient of environmental conditions to test the patterns of forest songbird distribution. *I predict that my research will corroborate that forest bird distribution is primarily driven by plot-scale characteristics of the forest vegetation. Furthermore, I predict that plots with mid-successional vegetation types and high vertical complexity will have the highest bird species richness and abundance (H5a).*

<u>5.2 Keystone structures</u>

A "keystone structure" is a distinct spatial structure providing resources, shelter or 'goods and services' crucial for other species (Tews et al. 2004), and shows a positive relationship between habitat heterogeneity and biodiversity. For example, forest gaps are small disturbances that disrupt the forest canopy, thereby altering the micro-environment and the availability of resources in the ecosystem. A variety of organisms otherwise excluded from the closed-canopy forest can respond to the variable resource conditions created by gaps. Savannah trees serve a similar role in savannah ecosystems because, through their distinct structure, they provide a set of micro-climatic conditions and resources otherwise lacking in the environment.

Is a forest gap simply an island in a sea of closed-canopy forest? Are savannah trees simply an oasis of shade, food, and moisture in an inhospitable stretch of grassland? Both forest gaps and savannah trees are structures embedded in an otherwise homogeneous landscape, and are essential for ecosystem functioning and diversity. Although small in proportion to the entire ecosystem, keystone structures are essential to maintain biodiversity at the landscape scale because their presence adds heterogeneity, ensuring greater species diversification and coexistence. Thus, rather than perceive forest gaps and savannah trees as habitat islands, it may be more appropriate to perceive them as keystone structures for the landscape as a whole.

Sacred forests are patches of native forest that are surrounded by homogeneous stands of pine and oak. This raises the question of whether sacred forests are simply islands of habitat surrounded by a degraded and homogeneous matrix, or do they function as “keystone structures” that are embedded within and integral to maintaining diversity at the landscape scale?

There are important characteristics of the Tibetan sacred forest that indicate its similarity to a keystone structure in both its structure and function. First, sacred forests are numerous and occupy a wide range of sizes (0.1-1000 km^2). They are located in a broad range of topographic and micro-climatic conditions, and harbor a variety of vegetation types. Thus, sacred forests, like forest gaps and savannah trees, are small, numerous, irregular, and dispersed throughout the landscape, creating a great deal of between-site heterogeneity. Second, sacred forests experience a gradient of human disturbance that corresponds with distance from the focal point (e.g. temple, summit, altar) as well as topography. Thus, there is a great deal of within-patch heterogeneity. Within-patch heterogeneity creates an environment where a variety of organisms can respond to and thrive in the variable resource conditions of the patch (Belsky and Canham 1994). Finally, sacred forests are not separated from the matrix by an abrupt barrier. Rather, there is a continuous gradation of resources to levels characteristic of the surrounding matrix. The gradual blending of the patch with the matrix facilitates greater interaction between the patch and the matrix (Belsky and Canham 1994), and such interactions have a very important influence on a wide range of animal species (Prugh et al. 2008). Thus, *I hypothesize that sacred forests are a keystone structure for avian diversity in the Tibetan landscape (H5b).*

5.3 Patch/Matrix Dynamics

Increasing habitat fragmentation eventually results in a state where habitat occurs only in patches, surrounded by a non-habitat matrix. Historically, ecologists studied remnant habitats in terrestrial ecosystems as isolated and discrete patches, but it is becoming increasingly evident that patches do not necessarily function according to the principles of island biogeography (Prugh et al. 2008). For example, traditional fragmentation measures, such as patch area and isolation, are important for bird species diversity in native forests surrounded by an agriculture-dominated landscape (Opdam et al. 1985; Daily et al. 2001). However, where the matrix is relatively similar to the fragments (e.g. native forest surrounded by pine plantation), patch area or isolation *are not* strong predictors of bird diversity or abundance (Estades and Temple 1999; Bhagwat and Rutte 2006). In many ecosystems, the predictions of island biogeography do not hold true because animal movement and behavior is highly influenced by the composition of the matrix (Ricketts 2001; Prugh et al. 2008). In NW Yunnan, sacred forest plots are mostly surrounded by homogeneous secondary forest, which is not completely unsuitable habitat for birds. *Thus, I hypothesize that patch/matrix interactions are very important for bird communities in NW Yunnan, and that environmental conditions surrounding the patch, i.e. the quality and composition of the matrix, influence bird abundance and diversity (H5c).*

Technical Approach

Sacred Forest Site Selection and Sampling Design

I visited several sacred forest sites during fieldwork in Fall 2008. I hired a local Tibetan driver and his car to take me to sacred forests known to him. I briefly hiked in each site (1-3 hours). From these several forests I have selected 6 as potential research sites (Table 6, Figure 15).

During Fall 2009, I will revisit these 6 sites, and ask for permission from the villagers to use their forest as a research site. If they agree, I will hire a guide from the village to accompany me to the sacred forest to learn the following: 1) the location of the focal point of the sacred site, 2) the approximate boundaries of the sacred forest, and 3) paths within the forest that I could use to access sample plots. I will ask my guide general questions about the management of their forest, because village-specific management clearly has an influence on plant and animal communities. For example, logging removes overstory trees, firewood harvest threatens both canopy and understory vegetation, grazing threatens understory diversity and seedling development, and hunting directly threatens bird abundance and diversity. Such information will increase understanding of the variability in the meaning and management of sacred forests across several Tibetan villages.

Breeding bird surveys will be conducted twice during the breeding season (May through June) of 2010 at 6 sacred forests (sites). At each site, I will survey approximately 12 plots, 6 within the sacred forest and 6 in the surrounding forest matrix (Figure 16), for a total of 36 sacred and 36 matrix sites. Plots outside of the sacred forest will be randomly selected within zones of distance (within 50 m, 51-250 m, and 251-500 m) from the sacred forest boundary. Plots inside the sacred forest will follow an existing walking path, and plots along these paths will be randomly selected based on zones of distance from the sacred forest focal point (e.g. near, intermediate, and far; exact distances will depend on the size and dimensions of the sacred forest). Point counts will be performed within sites of 50 m radius according to standardized methods (Ralph et al. 1993), and will include an estimate of distance to bird using a hand-held laser rangefinder, for later density calculation (Buckland et al. 1993). There will be a minimum distance of 100 m between plots. Dominant plant species in each life form class (tree, shrub, grass and forb) as well as foliage height diversity will be recorded at each plot.

Table 6. Six sacred forest proposed as study sites.

ID	Area (ha)	Aspect	Elevation	Distance to Shika (km)	Description
SF1	20	South and East	3300-3600	1.3	Native shrubs in a rocky gorge.
SF2	8	Hilltop and sides	3300-3400	2.2	Regenerating and mature native forest and shrub on a small hilltop. White birch, oak, rhododendron, spruce.
SF3	50	East	3300-3700	0.7	Mature and secondary native forest near village. Birch, oak, spruce/fir, pine.
SF4	48	West	3300-3600	8.8	Near city of Zhongdian, heavily visited. Larch, oak, and rhododendron
SF5	35	Northwest	3200-3500	14.8	Mature pine, oak shrub, spruce, bamboo, ash.
SF6	25	North	3300-3600	21.8	Regenerating mixed forest with mature trees at top of ridge

Figure 15. Six proposed sacred forest sites and Shika Snow Mountain, a sacred mountain whose surrounding forest is also considered sacred. Refer to Table 6 for Site IDs and descriptions. Boundaries are approximate and estimated from personal observation and high-resolution images on Google Earth.

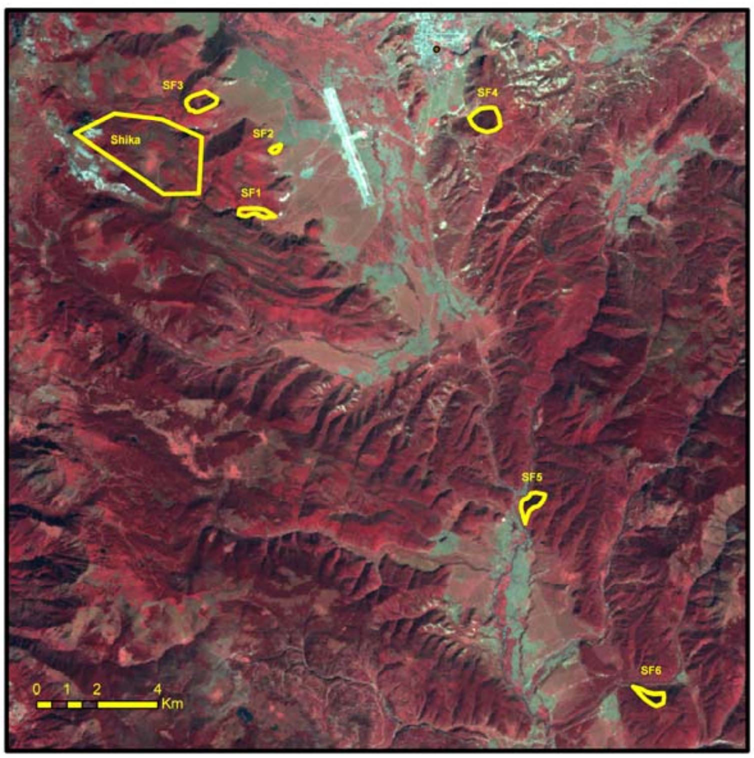


Figure 16. Example of sample design at SF4. Plots labeled with numbers represent plots within the sacred forest. Plots labeled with letters represent plots in the matrix. Wufengshan has a small Buddhist temple that is its focal point. Boundaries of the sacred forest (in red) are approximate and estimated by personal observation. Wufengshan is very near (1km) the capital of Shangri-la County, and thus the matrix forest surrounding Wufengshan is intensively used and quite degraded.

<u>Analysis</u>

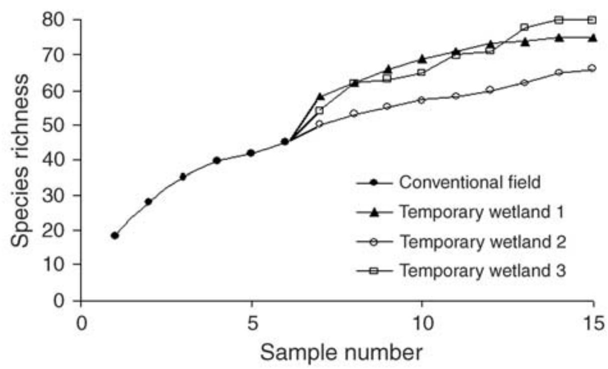
5.1 Habitat selection of Forest Songbirds

Sacred forests exhibit a great deal of between and within-site variability. Therefore, although I am visiting a small number of sites (six), I will be collecting data for a relatively wide range of forest types and successional stages. For my first stage of statistical analysis, I will not stratify the samples into forest types or classes. I will consider that the 60 plots represent a gradient of vegetation heterogeneity, including vertical structure, horizontal structure, and species diversity. I will build a model (e.g., multiple regression) to predict avian species abundance and diversity with measures of vegetation heterogeneity, topography, and its location within or outside of a sacred forest. For example, I predict that avian diversity and abundance will be driven primarily by plot-scale measures of vegetation heterogeneity. This analysis will serve to 1) provide a description of the bird communities that are present within the region, and 2) demonstrate overarching habitat factors driving forest bird species abundance and diversity.

5.2 Sacred forests as a keystone structure

Next, I will use the survey data to determine whether sacred forests can be considered a keystone structure. I will compare vegetation structure within versus outside of sacred forests to determine whether sacred forests are features that add heterogeneity to the landscape. I predict that plant communities within sacred forests are highly diverse in terms of type and structure, compared to the more homogeneous matrix vegetation. I further hypothesize that this increased heterogeneity provides a greater variety of resources that in turn drives an increased bird abundance and diversity within sacred forests. Tews et al. (2004) consider that a primary indicator of a keystone structure is that there are abrupt discontinuities in the species accumulation curves when a sampling transect enters the structure (Figure 17). I predict that species-accumulation curves at each site will demonstrate abrupt increases when entering the sacred forest patches. I will demonstrate this same concept statistically using paired comparisons tests. I predict that bird diversity and abundance will be significantly higher in sacred forests than in the matrix at each site.

Figure 17. Temporary wetlands as keystone structures for carabid beetles in a conventional field. Abrupt discontinuities in species-accumulation curves occur as sampling transects enter a temporary wetland (from Tews et al. 2004)



5.3 Patch/Mosaic Dynamics

I hypothesize that sacred forests do not function as isolated islands. To test this, I will include traditional measures of habitat fragmentation (e.g. patch size and distance to another sacred forest) in the statistical model described in 5.1. For example, I predict that patch size will not influence overall measures of species richness and abundance within sacred forests. However, I do predict that patch size will influence the bird community composition, because species that require large areas of native forest will be sensitive to patch size, and thus absent from smaller patches.

Patch/Matrix Dynamics at the Patch Scale

In addition to within-site characteristics, interactions between the patch and the *immediate* matrix will influence bird abundance and diversity. For example, the secondary pine/oak matrix is probably not impenetrable to most birds, and may provide resources that complement those within the sacred forests. Therefore, the quality of the matrix probably influences within-patch bird communities. For example, I predict that within-patch bird abundance and diversity will be influenced by measures of vegetation characteristics of the matrix plots because extremely degraded matrix forests will provide less niche space overall (e.g. fewer suitable nest sites and fewer foraging resources). However, my sample size (n = 6) may not be sufficient to adequately address this question.

I predict that sacred forests also influence the matrix bird communities. Sample plots will be located outside of the sacred forest based on their distance from the patch boundary. Matrix vegetation tends to be quite homogeneous, and therefore I predict that there will be only small differences in the vegetation measures between matrix plots. However, I predict that bird abundance and diversity will be higher closer to sacred forest patches than far away. The edge between the patch and the matrix is not a discrete boundary, thus facilitating interaction in this transition zone. Breeding birds that live within the sacred forest likely use and move through areas of the matrix that are close to the sacred forest more than areas further away. Likewise, sacred forest resources are more accessible from the matrix near to the patch than far away from the patch.

Patch/Matrix Dynamics at the Landscape Scale

The sites are naturally stratified to allow exploration of whether the distribution of patches on the landscape affects bird communities. Three of the sites are clustered together (Figure 15; SF1, SF2, and SF3) and close to a sacred mountain (Shika Snow Mountain), which protects a large (~250 ha) sacred forest (Shika). SF1, SF2, and SF3 will represent “high-density” sacred forest sites, because there is a relatively high density of natural forest (i.e. other sacred forests and Shika Snow Mountain) in their landscape. The three other sites are more dispersed and isolated (SF4, SF5 and SF6). These three sites will represent “low-density” sacred forest sites, because there is a relatively small proportion of natural forest at the landscape scale (i.e. a single patch amidst a pine/oak matrix). I predict that bird abundance and diversity will be higher in high-density sites for two reasons. First, the three study sites plus Shika Snow Mountain are in close proximity to each other, resulting in a higher proportion of high-quality habitat at the landscape scale. Second, the high-density sites are in close proximity to a large forest patch, Shika Snow Mountain, which may act as a ‘source’ for individuals dispersing from high-quality habitat to less densely populated habitat (Pulliam 1988).

Chapter 5: Expected Outcomes and Significance

Findings of this chapter will be summarized in a manuscript submitted to Ecological Applications. It will discuss general characteristics of sacred forests in the Tibetan landscape, including their structure, meaning, and pattern on the landscape. The manuscript will also describe the avian forest breeding bird communities in the study area, and the important features that drive habitat selection. The manuscript will also explain the idea of keystone structures in general, and discuss whether sacred forests should be considered a keystone structure for avian biodiversity in the Tibetan landscape. Finally, the paper will add to the body of existing literature that questions whether the “island” or “mosaic” conceptual framework is a more appropriate model to study habitat patches in terrestrial environments.

In general, the ideas of keystone structures and patch/matrix interactions have important implications for conservation and land management world-wide. Viewing remnant habitats as integrated and essential components of the landscape, rather than isolated patches, has an important advantages. Namely, recognizing individual patches as part of a landscape-scale network that is essential to maintain high biodiversity and ecosystem functioning emphasizes the need to protect even the tiniest of patches, and provides focus to direct limited resources.

Finally, this research will have special relevance for conservationists and policy-makers in SW China. The research adds to our very limited knowledge about 1) avian communities in NW Yunnan and 2) the meaning, use, and distribution of sacred areas. Furthermore, the research takes place in a particular region of SW China where biodiversity is already threatened, and where economic development and cultural change will continue in the next decades. Tibetan culture will surely evolve, but it is unlikely that local Tibetans will forgo subsistence activities in the forest matrix in the near future. However, if the culture and beliefs that already maintain sacred forests are validated and supported, these patches will likely persist on the landscape even in the face of economic development and cultural change.

OVERALL SIGNIFICANCE OF THE DISSERTATION

My research will be broadly relevant to remote sensing, land use science, and conservation biology (Table 7). NW Yunnan provides a fascinating case study because it is a unique and unstudied region that is quite distinct from other biodiversity hotspots in developing regions. NW Yunnan contains mountainous, temperate ecosystems not found in Africa or Latin America. China has a strong central government and ability to effect change through policy and regulation. Furthermore, studying ecosystem change patterns and processes in an area experiencing accelerated processes of global climate change gives us an opportunity to gain insights to future dynamics in other regions.

My research will contribute to the fields of Remote Sensing Science, LULCC Science, and Conservation Biology in three ways. First, I will make advances in **technical approaches** that will make the research relevant to a wider remote sensing audience. I will use multi-temporal image analysis to overcome some of the challenges inherent for remote sensing analysis of mountainous ecosystems with a monsoonal climate. In addition, I will compare two very different LULCC classification methods, a hybrid parametric method versus Support Vector Machines (SVM), a machine-learning algorithm only recently adapted for multi-class image analysis. In particular, I will test the ability of SVM to tackle multi-class problems with non-random training data. Finally, my research will explore a new method of trajectory analysis, using non-anniversary date images to detect forest disturbance, which would be an exciting step towards fully exploiting NASA's recently released Landsat archive record.

Table 7. Proposed manuscripts from the dissertation.

Chapter	Key Question	Journal
1	What are regional patterns of LULCC?	Remote Sensing of Environment
2	Have shrubs encroached into alpine meadows at the regional scale?	Remote Sensing of Environment
2	Does the burn ban reduce heterogeneity of alpine ecosystems at multiple spatial scales?	Landscape Ecology
3	Has the logging ban successfully protected high-diversity forests?	Remote Sensing of Environment
4	Have government-sponsored protected areas or Tibetan sacred forests been more effective?	Conservation Biology
5	What is the role of Tibetan sacred forests to maintain avian biodiversity?	Ecological Applications

My dissertation, because it takes place in two unique and relatively unknown ecosystems (Himalayan temperate forests and alpine ecosystems), advances **ecological** knowledge in two ways. First, gathering baseline information on unstudied ecosystems is important in and of itself, but also a necessary first step on which to base future ecological research. Second, I will test ecological theories in a unique environment. For example, Himalayan alpine meadows are very diverse, but we do not understand the processes that create such high biodiversity. This research serves a dual purpose in that it, 1) documents post-fire succession of alpine shrubs, which is currently unknown, and 2) explores the role of patchy fire disturbance in alpine meadow ecosystems at multiple spatial scales. My research on forest birds also serves a dual purpose. It describes avian communities and their habitat selection in NW Yunnan forests, which has not yet been done. In addition, the research tests whether the "island" or "mosaic" conceptual framework is more appropriate to study habitat patches in terrestrial environments.

My research is also highly relevant for **conservation** in Southwest China, and around the world. The research examines three major environmental policies implemented by the Chinese government to protect the environment, the burn ban, the logging ban, and protected area establishment. A sufficient length of time has passed to determine the on-the-ground effects of these well-intentioned policies. In the case of the logging ban, the analysis of forest disturbance will allow us to understand how local people responded to the ban and the implications of the resulting landscape patterns for biodiversity. In the case of the burning ban and protected areas, my research addresses a wider issue. Human land use is usually assumed to be harmful for biodiversity, but I hypothesize that traditional Tibetan yak herding may serve to enhance biodiversity. Well-intentioned governmental policies may in some cases alter traditional forms of land management, with serious negative consequences for biodiversity.

The research has further contributions to conservation by analyzing two different kinds of protected areas in the same region, government-sponsored protected areas and Tibetan sacred forests. A major expansion (almost a doubling) of the state-sponsored protected area system is planned in the near future, and my regional analysis of existing protected area effectiveness comes at a critical time. I will also determine the contribution of sacred forests to conservation by testing their resilience over time, as well as their role in maintaining avian biodiversity in the region. As the region develops and minority cultures evolve, the acknowledgement of sacred forests as legitimate and important components of the landscape may facilitate their persistence on the landscape.

TIMELINE

	2009	**2010**				**2011**			
Activity	**Fall**	**Winter**	**Spring**	**Summer**	**Fall**	**Winter**	**Spring**	**Summer**	**Fall**
Chapter 1									
Fieldwork	x								
Image Analysis		x							
Writing		x							
Chapter 2									
Fieldwork	x								
Image Analysis					x				
Writing						x			
Chapter 3									
Fieldwork	x								
Image Analysis						x			
Writing							x		
Chapter 4									
Fieldwork	x		x	x					
Data Analysis								x	
Writing									x
Chapter 5									
Fieldwork	x		x	x					
Data Analysis					x				
Writing						x			
Defend Dissertation									x

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