



# The relative effectiveness of protected areas, a logging ban, and sacred areas for old-growth forest protection in southwest China



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## ABSTRACT

Forests are critically important for life on earth, prompting a variety of efforts to protect them. Protected areas and logging regulations are the most commonly used forest conservation strategies, but local traditions and religious beliefs can also protect natural resources by limiting exploitative use. We compared the effectiveness of protected areas, a logging ban, and sacred areas to protect forests from logging in Northwest Yunnan, China, a global biodiversity hotspot. We combined Mahalanobis matching and panel regression techniques to measure effectiveness of these three protection strategies paying special attention to old growth forest communities. We found that protected areas had no impact on total forest cover, but effectively conserved old-growth forests relative to non-protected areas. The implementation of the logging ban resulted in positive forest conservation outcomes over most of the landscape. The exception was that logging in old-growth forests inside sacred areas accelerated following the implementation of the logging ban, suggesting that local institutions may have been weakened by official policies. Our research finds little evidence that overlapping conservation policies decrease deforestation and suggests that the implementation of official policies may displace local forms of protection. Our results further highlight that relying on total forest cover as a single indicator of conservation outcomes can lead to misleading conclusions about the impacts of forest protection strategies.

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## 1. Introduction

Forests provide indispensable ecosystem services, including carbon sequestration, climate regulation and recreation, and are essential for biodiversity conservation (Kinzig et al., 2011). Their importance has prompted a wide range of efforts to protect them, including protected areas, logging regulations, and payments for ecosystem services (PES) programs (Brandt et al., 2012; Morell, 2008; Nepstad et al., 2011). The implementation of protected areas, where logging is legally restricted, is a common governmental policy tool worldwide (Bertzky et al., 2012). While 12.7% of the terrestrial earth is now protected, protection efforts have had mixed effects, especially in countries with weak governance or rampant growth in population and consumption (Ewers and Rodrigues, 2008; Knorn et al., 2012; Liu et al., 2001). Other government-sponsored forest protection measures include

prohibitions on logging, including full or partial bans on timber harvesting (Margono et al., 2012; Sarker et al., 2011). However, conflicts with local people or lax enforcement can limit the effectiveness of logging bans (Brandon and Wells, 1992; Ferraro et al., 2011).

Community-level institutions can be effective alternatives to governmental forest protection efforts (Bray et al., 2003; Nagendra et al., 2008). In particular, many traditional cultures respect “sacred areas” where logging is prohibited for religious reasons (Allendorf et al., 2014; Bhagwat and Rutte, 2006; Verschuren et al., 2010). Sacred areas act as *de facto* protected areas in traditional landscapes, are often effective at protecting forests, and are resilient to social change (Brandt et al., 2013; Dudley et al., 2009). However, there are examples of recent degradation of sacred areas, as cultural change, globalization, and national policies alter local forms of land management (Ostrom, 1990; Rutte, 2011).

To identify effective conservation solutions, it is crucial to (a) rigorously assess the impacts of alternative forms of protection and (b) examine if there are interactions among them. These issues

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gain special relevance in the context of the Northwest (NW) Yunnan Province, China. NW Yunnan is both a biodiversity hotspot (Myers et al., 2000) and a UNESCO world heritage site. NW Yunnan's natural forests, the most diverse temperate forests in the world (Morell, 2008), were decimated by government logging companies from the 1950s through the 1980s. Since 1990, multiple forest protection strategies, including protected areas, a logging ban, and community-managed sacred areas, have been concomitantly in effect.

Our goal was to compare the effectiveness of these different protection strategies to conserve old-growth forest cover. We created a spatial dataset of protection strategies by delineating boundaries of protected areas, logging ban areas, and Tibetan sacred areas. We quantified forest cover types (old-growth forest and pine forests) and changes among them based on Landsat satellite images over two decades: 1990–1999 and 1999–2009. Simply comparing changes in forest cover over different tenure systems and over time is not sufficient to assess the relative effectiveness of different protection strategies, because the areas under the different protection strategies are in almost all cases inherently different in other aspects as well (Andam et al., 2008; Joppa and Pfaff, 2010, 2009). For example, in our setting, sacred areas have higher levels of old growth forests, and are at higher elevations and have steeper slopes than protected areas or logging ban areas. Given these differences in initial conditions, they experience different demands for forest resources.

In order to control for underlying differences among different protection strategies, we used a 2-step modeling procedure. First, we used matching statistics to create datasets consisting of observations in each protection category that had similar observable characteristics at the beginning of our study period (1990) (Caliendo and Kopeinig, 2008). However, matching only provides bias-free estimates when all factors that influence both protection strategy designation and deforestation are included as controls, an assumption which cannot be tested. Thus, we next also applied

panel regression techniques to account for unobserved heterogeneity and identify the impacts of the different protection strategies across forest types. We asked three specific questions:

1. Did protected areas protect all forest types equally?
2. Did the region-wide logging ban effectively reduce logging compared to other protection strategies?
3. Did sacred areas continue to protect old-growth forests, even as new policies restricted logging in the surrounding landscape?

## 2. Methods

### 2.1. Study area

Our study area (20,036 km<sup>2</sup>) was the Diqing Tibetan Autonomous Prefecture in the Hengduan Mountains in Northwest Yunnan Province, bordering Tibet and Sichuan Province (Fig. 1). Northwest Yunnan is a biodiversity hotspot (Myers et al., 2000) with elevations ranging from 1500 to 6000 m above sea level, creating a high diversity of ecological niches in a relatively small area. Northwest Yunnan is also a UNESCO world heritage site, with great cultural diversity and at least 15 different ethnic minority groups in the region. Northwest Yunnan is still relatively undeveloped compared to other parts of China, but is experiencing rapid change. Since the 1970s, NW Yunnan has undergone major changes due to national policies aimed at fostering both economic development and environmental protection. These policies stimulated rapid infrastructure development, immigration of culturally-dominant Han Chinese, tourism, new protected areas, and changes in land use. Local peoples continue to practice subsistence-based agriculture and pastoralism, but livelihood strategies and traditional land use practices are evolving rapidly with increasing climate change and economic development.

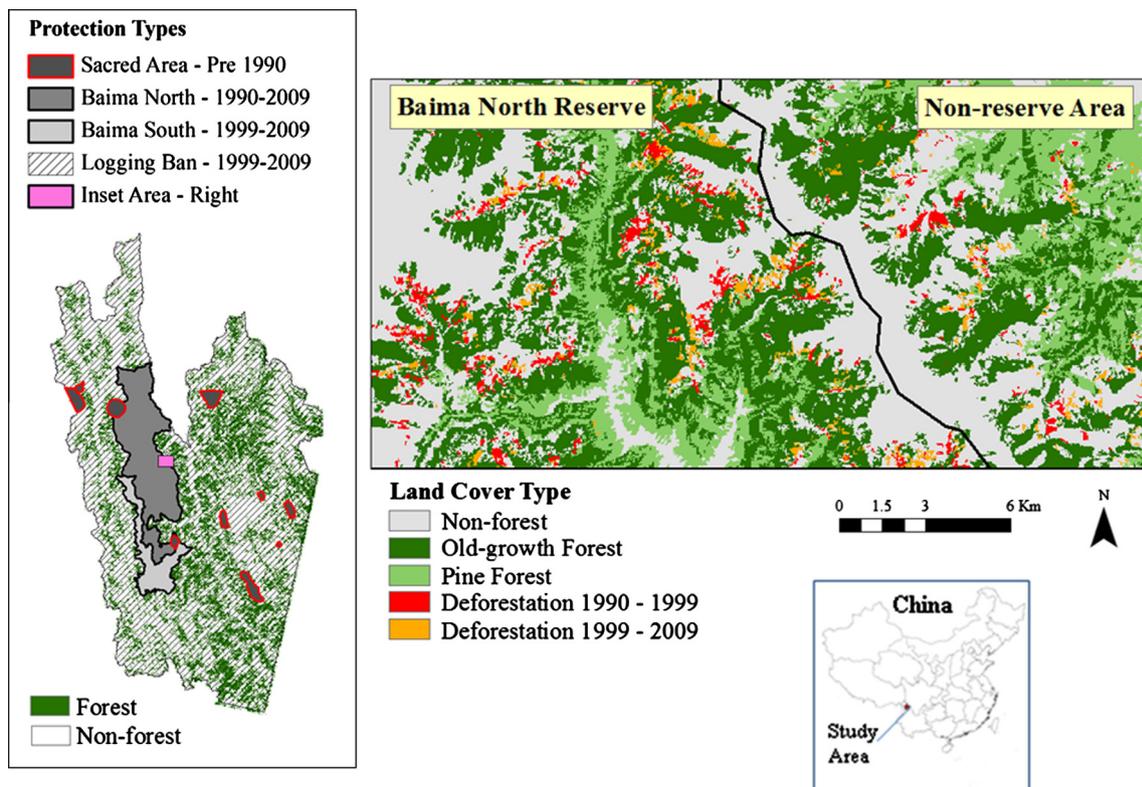


Fig. 1. Land cover and protection types in the study area in Northwest Yunnan Province of southwest China.

Our study area retains high levels of forest cover (>60%), but old-growth forests, the primary conservation target in the region, are only a fraction of that overall forest cover. Among all land cover types, old-growth forests have the highest levels of endemic and culturally useful species, and is crucial habitat for the endangered snub-nosed monkey and several threatened pheasant species (Ma et al., 2007; Wang et al., 2008). From the 1950s through the 1980s, the region's old-growth forests were heavily exploited by government logging companies. Since the 1990s, several forest protection policies have been enacted to protect old-growth forests, and since the national logging ban was implemented in 1998, all of our study area has been under some form of protection (Wang et al., 2007). Despite protection, old-growth forest logging accelerated in areas with a growing tourism industry, population growth and economic development (Brandt et al., 2012).

In light of continued socio-economic changes occurring in northwest Yunnan, it is important to determine the relative impact on forest ecosystems of the three different protection strategies in place: government protected areas, a logging ban, and Tibetan sacred areas. The government protected area in place is Baima Nature Reserve, one of southwest China's largest and oldest national-level protected areas, established to protect old-growth forests that are critical habitat for the endangered Yunnan Snub-nosed Monkey. The northern half of Baima Reserve (Baima North) was established in 1990; the southern part (Baima South) was added in 1999. Commercial logging has been banned throughout the reserve; local communities can log on a quota basis.

Next, China implemented a region-wide logging ban (i.e. the Natural Forest Protection Program and the Grain to Green Program) in 1998, in response to severe flooding of the Yangtze River in 1997 and 1998. These two programs banned all commercial logging in southwest China, and provided incentives to local farmers, governments and forest enterprises to reforest degraded lands. Together the two programs invested over US\$15 billion for forest protection and afforestation, making it the largest PES program in the world (Liu et al., 2008).

Finally, ethnic minorities of southwest China practice a traditional form of protected area, i.e., "sacred areas". In our study area, Tibetan sacred areas are most prominent. Typically, religious rituals are carried out at these sites, and extractive activities, such as grazing, cutting, hunting and agriculture, are prohibited. Tibetan sacred areas have been effective forms of protection even during periods of heavy logging, protecting remnant native forests even in highly impacted landscapes (Salick et al., 2007; UNESCO-MAB, 2003; Xu et al., 2005). Sacred areas have played an important conservation role throughout the Himalaya, because although typically small in size, they are scattered throughout the landscape and thus provide protection for a wide range of ecological niches, are resilient to intensive land use pressures, and play important roles as local refugia for native species (Bhagwat and Rutte, 2006; Brandt et al., 2013; Dudley et al., 2010; Shen et al., 2012).

## 2.2. Data

We analyzed a spatial dataset of unique observations for which we summarized (a) forest cover in three time periods, (b) the form of protection, and (c) social and environmental correlates of logging and protection strategy. We projected all datasets into Transverse Mercator projection and aggregated outcomes, treatment boundaries and control variables data into 200 m cells. Our study area contained >2 million  $200 \times 200$  m<sup>2</sup> cells of which ~400,000 were >90% forested. We analyzed only those cells that were >90% forested cells (old growth + pine forest) in 1990, to focus on the most important forests for conservation. We used cells that were 1000 m apart, a distance beyond which there was no significant spatial autocorrelation in the residuals of our regression model.

### 2.2.1. Forest cover

We used three measures of forest cover: old-growth forest, pine forest and total forest (old growth + pine) for three time periods (1990, 1999, 2009). Land cover/land use data was generated from Landsat MSS/TM/ETM + satellite imagery from 1990, 1999, and 2009 classified into eight classes: old-growth forest community, pine/oak woodlands, non-pine shrub and scrub, agriculture, grassland, bare land, snow, and unclassified land cover classes (Brandt et al., 2012). Our old-growth forest community class represents the native forest vegetation community in both its climax and regenerating (i.e., secondary) state (Li and Walker, 1986), covers ~20% of the landscape, and is the main conservation target in this region. The old-growth forest community contains mixed evergreen and deciduous species including fir (*Abies* spp.), spruce (*Picea* spp.), pine (*Pinus* spp.), larch (*Larix* spp.), evergreen oak (*Quercus* spp.), birch (*Betula* spp.) and rhododendron (*Rhododendron* spp.). Pine forests are typically homogeneous secondary forests of *Pinus densata* with oak shrub (*Quercus* spp.) understory. Pine forests have regenerated after logging and cover ~20% of the landscape. Pine forests generally have low biodiversity value compared to old-growth forests, but provide essential resources (i.e. construction materials and fuelwood) for local people (Brandt et al., 2012; Zhou and Grumbine, 2011). All areas not classified as old-growth or pine forests were considered non-forested for our analysis.

### 2.2.2. Protection type

We evaluated impacts of three different forms of forest protection (Fig. 1, Table 1): (a) government-sponsored protected areas, (b) the region-wide logging ban, and (c) sacred areas. We had two different reserves implementations in our study area. The northern half of Baima Reserve (Baima North) was established in 1990; the southern part (Baima South) was added in 1999. Reserve boundaries for Baima North and Baima South were obtained from the Yunnan Forestry Bureau. The logging ban prohibits all commercial logging in southwest China, and thus effectively covers our entire study area in the 1999–2009 time period (Liu et al., 2008).

**Table 1**

List of protection types, year of policy implementation, policy relevant questions, comparison groups and methods.

Protection type	Year policy started	Relevant policy question	Comparison groups (years relevant)
Baima North <sup>a</sup>	1990	Are PAs effective relative to areas with no protection policy?	Baima North vs. logging ban (1990–1999) Baima North vs. Baima South (1990–1999)
Logging ban <sup>a</sup>	1999	Is the logging ban effective relative to PAs?	Logging ban vs. Baima North (2000–2009) Logging ban vs. Baima South (2000–2009)
Sacred areas <sup>b</sup>	Pre-1990	Are sacred areas effective relative to areas with no protection policy?	Sacred areas vs. Baima South (1990–1999) Sacred areas vs. logging ban (1990–1999)
		Are sacred areas effective relative to PAs?	Sacred areas vs. Baima North (1990–1999, 2000–2009) Sacred areas vs. Baima South (2000–2009)
		Are sacred areas effective relative to the logging ban?	Sacred areas vs. logging ban (2000–2009)

<sup>a</sup> Modeling method: Mahalanobis matching and post-matching fixed effects difference in differences regression.

<sup>b</sup> Modeling method: Mahalanobis matching and post-matching fixed effects regression.

**Table 2**

The area (km<sup>2</sup>) and proportional coverage (percent) of three different forest types in each protection category in the three time periods (1990, 1999, and 2009).

Year	Baima North		Baima South		Logging ban areas		Sacred areas	
	Area (km <sup>2</sup> )	% Cover						
<i>Old-growth forest</i>								
1990	745	35	226	33	4307	26	168	44
1999	672	31	214	32	3583	21	148	39
2009	622	29	202	30	3477	20	131	35
<i>Pine forest</i>								
1990	332	15	199	29	4235	25	13	3
1999	347	17	210	31	4422	27	11	3
2009	313	15	188	28	4225	25	11	3
<i>Total forest (old-growth + pine forest)</i>								
1990	1077	50	425	63	8542	51	181	48
1999	1019	47	424	62	8005	48	160	42
2009	935	43	390	58	7702	45	142	38

Boundaries of sacred areas were collected in the field by talking to villagers and from the literature (UNESCO-MAB, 2003).

### 2.2.3. Control variables

We generated a spatially-explicit data set of 12 variables which are commonly thought to influence the likelihood of forest protection as well as forest harvesting. These included: (1) road density, (2) road density squared, (3) distance to Shangrila, (4) elevation, (5) elevation squared, (6) slope, (7) slope squared, (8) % edge forest, (9) % core forest, (10) % old growth forest, (11) % pine forest, and (12) % snow. We obtained spatial data on road networks, provincial boundaries and major cities from 1:250,000 topographic maps from the late 1990s. We used the length of roads (including national, provincial, county and village-level roads) within each township to measure road density. We calculated slope and elevation from a 90 m-resolution Digital Elevation Model (DEM) recorded by the Shuttle Radar Topography Mission (SRTM). Percent core and edge were calculated using Guidos software (Vogt et al., 2007) (See SI for full description of variables and rationale for including them).

### 2.3. Identification strategy

To identify the relative impacts of various protection strategies, we implemented a two-step modeling approach. First, we were concerned that protected areas and sacred areas were incomparable to non-protected areas due to differences in underlying characteristics (Joppa and Pfaff, 2009). This may create sample selection bias that can translate into biased regression models (Andam et al., 2008). We therefore used Mahalanobis matching to select observations that had similar observable control variables, but different types of protection (Caliendo and Kopeinig, 2008). We applied these matching techniques to our first year of data, 1990, which pre-dated formal conservation policies in our study area. These matched observations were then saved as individual datasets, merged with data from subsequent time periods and used in a second stage regression framework.

Our matching approach had to consider that we were interested in four categories of treatment (Baima North, Baima South, logging ban, and sacred areas). Therefore, we created six matched datasets (Baima North vs. logging ban, Baima South vs. logging ban, Baima North vs. Baima South, Baima North vs. sacred areas, Baima South vs. sacred areas, and logging ban vs. sacred areas) which enabled us to test the effectiveness of each protection strategy by comparing it to every other strategy. We test for the quality of our matches by calculating the amount of bias reduced between matched and unmatched pairs (SI Tables 1–6), as well as by comparing the

empirical quantile-quantile (eQQ) differences of each matched and unmatched set (SI Tables 7–8). Unclassified land was the excluded land cover category in the matching algorithms.

While matching can correct for selection bias created by differences in observable control variables, it is poorly suited to correct for bias due to unobserved heterogeneity in the data (i.e. omitted variable bias (Heckman et al., 1998)). The panel structure of our dataset gave us the ability to control for some sources of omitted variable bias. Our strongest identification strategy was a fixed-effects difference-in-differences (DID) model, which relies on the variation across time *within* each cell. Such a model can correct for both time invariant and time-varying omitted variables, under the assumption that the time varying omitted variables in both the treated and non-treated observations have similar trends (Cameron and Trivedi, 2005). While the direct impact of time invariant variables are not estimated (for example slope and elevation) they are controlled for within the fixed effects framework.

Our basic post-matching regression equation takes the following form

$$FC_{it} = \delta(Treat_{it}) + X'_{it}\beta + \theta_t + \gamma_i + \varepsilon_{it} \quad (1)$$

where  $FC_{it}$  denotes the forest cover variable of interest,  $Treat_{it}$  is a binary variable equal to one if cell  $i$  is protected by the policy of interest in time  $t$ , and  $X_{it}$  are cell level characteristics that vary with time,  $\beta$  are coefficients to be estimated,  $\theta_t$  is a time specific impact,  $\gamma_i$  is an cell specific fixed effect, and  $\varepsilon_{it}$  is the random error component. Omitting the control variables for simplicity, in the two period model estimated here, the parameter  $\hat{\delta}$  gives the DID estimator

$$\hat{\delta} = \overline{\Delta FC}_{treatment} - \overline{\Delta FC}_{control} \quad (2)$$

The first and second terms in Eq. (2) each account for the time-invariant attributes that affect the level of forest cover within each cell, even if these attributes are unobserved, and the difference between the terms accounts for trends that impacted both areas simultaneously. As long as no unobserved time variant trend impacted one group but not the other (i.e. the 'parallel trend' assumption), the DID estimator produces a valid estimate of the impact of the policy on the treated group. The fixed effects difference-in-differences model is similar in design to the before-after-control-impact experimental design, as it requires having observations both before and after the treatment is applied. In our study area, we have before and after data for both the establishment of the protected areas and for the logging ban, and can therefore use a DID model for any comparisons involving at least one of these protection strategies. For comparisons that involve sacred areas, we had no pre-policy observations, and therefore difference-in-differences modeling was impossible. For this comparison we use a fixed

effects panel regression model to estimate the relative changes in forest cover. This requires us to make the additional assumption that there are no time varying omitted variables in our data. Likewise, we must match these areas based on 1990 characteristics, though sacred areas existed before this time. Two additional factors may affect the impact estimates for sacred areas. First, there are few sacred mountain cells in our dataset compared to the other treatments, and thus it is possible that the harvest of relatively small areas may lead to a large estimate impact. Second, because of the small number of samples, it was necessary to use all available observations, so we were not able to use a spatial sample. All independent variables were checked for collinearity over 0.50, but none was found in our datasets. Please refer to SI Tables 9–16 for full regression results for each paired impact analysis, and SI Table 17 for summary statistics for key variables used in matching and regression.

### 3. Results

Of the three protection strategies, logging ban areas contained by far the largest total area of old-growth forest, with 4307 km<sup>2</sup> in 1990, declining to 3501 km<sup>2</sup> by 2009 (Table 2). Protected areas contained the second-largest area of old-growth forests. Together, Baima North and South had 971 km<sup>2</sup> of old-growth forest at the beginning of our study period (1990), which declined to 825 km<sup>2</sup> by 2009. Sacred areas in our study area were small relative to the other protection regimes, containing 168 km<sup>2</sup> of old-growth forest in 1990, which declined to 131 km<sup>2</sup> by 2009. In terms of proportional area of old-growth forest, sacred areas had the highest percent coverage, with 44% of sacred areas covered in old-growth forest at the beginning of our study period (1990) compared to only 34% of Baima Reserve and 26% of logging ban areas.

#### 3.1. Protected areas effectiveness

We measured the impact of protected area implementation in the decade after Baima North's establishment (1990–1999) by comparing forest cover change in Baima North to areas on the landscape that were not yet protected by any policy (i.e., logging ban areas and Baima South). The impact of the policy can be interpreted as the relative change between Baima North and the control area (i.e., Change in Baima North cover – Change in control cover). Compared to logging ban areas, the establishment of Baima North led to no statistically significant change in overall forest cover. However, the impact of Baima North on old growth forest community cover was a relative change of 5.6% compared to logging ban areas (Fig. 2, Table 3). Compared to the area that would later become Baima South, there was a relative decrease in Baima North total forest cover, but old-growth and pine forest impacts separately were not significantly different from zero.

#### 3.2. Logging ban effectiveness

We measured the impact of the logging ban by comparing forest cover change in logging ban areas to protected areas (i.e. Baima North and Baima South) (Fig. 3, Table 3) in the decade following logging ban implementation. We found that the logging ban protected total forest and old-growth forest equally compared to Baima North, but protected pine forest better than Baima North (+3.2% relative change). Compared to Baima South, the logging

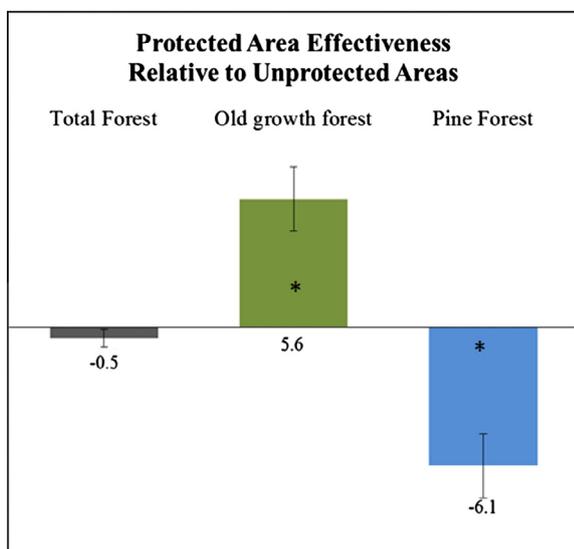


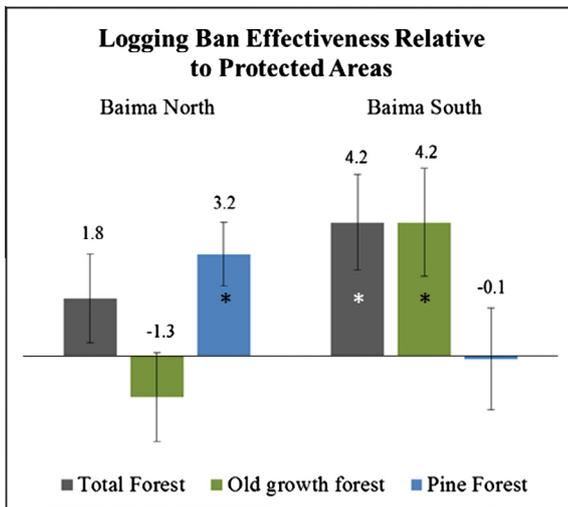
Fig. 2. Percentage point difference of total forest, old-growth forest, and pine forest in Baima North protected area relative to unprotected areas (i.e. logging ban areas in 1990–1999, prior to the implementation of the logging ban). Error bars show standard error of the mean calculated for each estimate. \* Indicates significance ( $p < 0.05$ ).

Table 3  
Summary of regression results showing relative effectiveness of different strategies to protect total forest cover, old-growth forest and pine forest.

Test	Percentage point change (standard error)		
	Total forest	Old growth forest	Pine forest
<i>Protected areas relative to areas without protection</i>			
Baima North vs. logging ban (pre-ban, 1990–1999)	-0.5 (.4)	5.6 (1.4) <sup>***</sup>	-6.1 (1.4) <sup>***</sup>
Baima North vs. Baima South (pre-Baima South, 1990–1999)	-1.7 (.5) <sup>***</sup>	1.2 (2.6)	-2.9 (2.7)
<i>Logging ban relative to protected areas</i>			
Logging ban vs. Baima North (1999–2009)	1.8 (1.4)	-1.3 (1.4)	3.2 (1.0) <sup>***</sup>
Logging ban vs. Baima South (1999–2009)	4.2 (1.5) <sup>***</sup>	4.2 (1.7) <sup>**</sup>	-0.1 (1.6)
<i>Sacred areas relative to protected areas and logging ban areas</i>			
Sacred areas vs. Baima North (1990–1999)	-2.7 <sup>***</sup> (0.7)	-0.9 (1.5)	-1.8 (1.4)
Sacred areas vs. Baima North (1999–2009)	-8.0 <sup>***</sup> (1.2)	-7.8 <sup>***</sup> (1.4)	-0.3 (1.1)
Sacred areas vs. Baima South (pre-Baima South, 1990–1999)	-2.1 <sup>***</sup> (0.7)	3.1 (1.5) <sup>**</sup>	-5.2 (1.3) <sup>***</sup>
Sacred areas vs. Baima South (1999–2009)	2.3 (2.5)	6.1 (2.6) <sup>**</sup>	-3.8 (1.2) <sup>***</sup>
Sacred areas vs. logging ban (pre-ban: 1990–1999)	.5 (0.5)	1.1 (0.6) <sup>**</sup>	-0.7 (0.3) <sup>**</sup>
Sacred areas vs. logging ban (post-ban: 1999–2009)	-0.7 (1.0)	-0.9 (1.0)	0.2 (0.3)

<sup>\*\*\*</sup>  $p < 0.01$ .

<sup>\*\*</sup>  $p < 0.05$ .



**Fig. 3.** Percentage point difference of total forest, old-growth forest, and pine forest in logging ban areas relative to protected areas in 1999–2009, after the implementation of the logging ban. Error bars show standard error of the mean calculated for each estimate. \* Indicates significance ( $p < 0.05$ ).

ban effectively protected total forest cover (+4.2% relative change), and old growth forest community cover (+4.2% relative change), and offered equal protection for pine forests. Thus, most evidence indicates that the logging ban was as effective, and in some cases even more effective, than the protected areas.

### 3.3. Sacred areas effectiveness

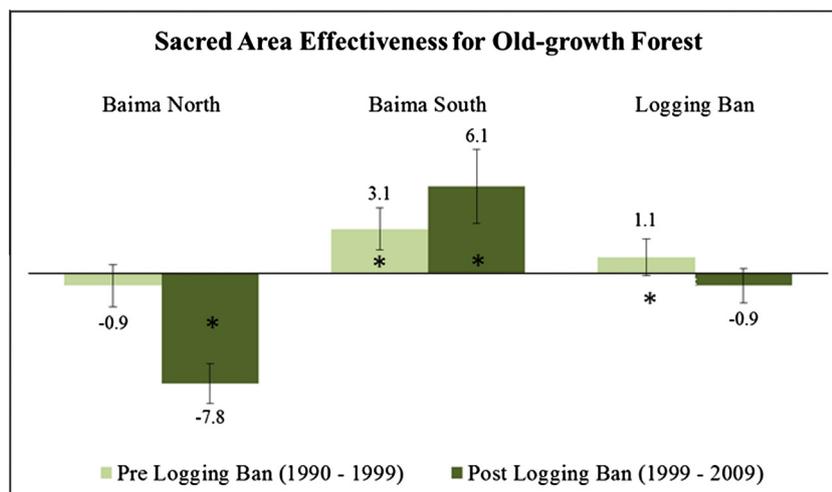
We measured the impact of sacred areas on old-growth forest by comparing old-growth forest cover in sacred areas to Baima North, Baima South and logging ban areas in each time period (Fig. 4, Table 3). Old-growth forest cover in sacred areas was equal to that in Baima North in 1990–1999, but effectiveness declined dramatically in the subsequent time period (–7.8% relative change). Sacred areas protected old-growth forests better than Baima South in both time periods: +3.1% relative change in 1990–1999, before Baima South was implemented, and +6.1% relative change in 1999–2009, after Baima South was implemented. Old-growth forest cover was higher in sacred areas than in logging ban areas prior to the implementation of the logging ban (+1.1% relative change), but

effectiveness declined and old-growth forest cover was equal in sacred areas relative to logging ban areas after the implementation of the logging ban. In summary, two of the three comparisons indicated that sacred areas decreased in effectiveness over time in our study period, as logging increasingly became restricted.

## 4. Discussion

Our results demonstrate that focusing only on overall forest cover can mask ecologically important heterogeneity in the effects of conservation policy. For example, we found no evidence for a significant impact of protected areas on preserving total forest cover, which would suggest that protected areas had been ineffective in our study area. However, when looking at forest types separately, we found that protected areas effectively protected old growth forests, and had a negative impact on secondary pine forests. These are important differences because of the biological and socio-economic value that these different forests hold. Old-growth forests are the primary biodiversity conservation target, and thus our results suggest positive outcomes. However, pine forest loss, although less important from a conservation perspective, are critically important to local people, and their decline has negative implications for livelihoods and ecosystem services (Li and Walker, 1986; Liu et al., 2008; Zhou and Grumbine, 2011). Many effectiveness studies consider only total forest cover as their indicator of conservation success (Andam et al., 2013, 2008; Honey-Roses et al., 2011), but our results suggest that relying on simple forest/non-forest assessments in environments with heterogeneous forest cover can obscure actual impacts of different conservation policies (Duque et al., 2014).

Our results indicate that the logging ban effectively protected forests in areas that were previously unprotected. The majority of timber harvest prior to the logging ban was performed by state logging companies, which were effectively disbanded once the logging ban was implemented, and this is one important reason for lower deforestation rates (Brandt et al., 2012; Zackey, 2007). The logging ban allows logging for local people for subsistence use on a strict quota basis. However, a burgeoning tourism industry, regional population growth and economic development have increased demand for forest products for both subsistence and commercial use (Brandt et al., 2012). We found that after the implementation of the ban, forests in protected areas and sacred areas fared no better, and in some cases worse, than areas



**Fig. 4.** Percentage point difference in old-growth forest in sacred areas relative to protected areas and logging ban areas in 1990–1999 and 1999–2009. Error bars show standard error of the mean calculated for each estimate. \* Indicates significance ( $p < 0.05$ ).

protected only by the logging ban. This means that overlapping conservation policies (i.e., logging bans on top of protected areas and sacred areas) did not necessarily increase forest protection throughout the study area, and in fact may have simply moved forest harvest from areas with one form of protection to another. Specifically, our results suggest that pine forests in protected areas, and old-growth forests in sacred areas, may be bearing the brunt of the increased pressure.

Sacred area effectiveness declined during our study period in relation to the rest of the landscape. At the beginning of our study period sacred areas retained high old-growth forest cover, even in highly impacted parts of the landscape, demonstrating that they have acted as effective *de-facto* protected areas for old-growth forests for centuries (Brandt et al., 2013; Salick et al., 2007). In the first time period of our study, sacred areas offered equal or better protection to old-growth forests compared to other regions, but after the establishment of both logging bans and parks, evidence on their effectiveness was mixed. Compared to Baima North and logging ban areas, old-growth forest loss accelerated in sacred areas after the logging ban, which suggests that the ban may have led to increased harvest within sacred areas. This result supports other findings where traditional forms of forest management and protection become weakened or displaced by the establishment of formal protection (Ostrom, 1990; Rutte, 2011; Verschuren et al., 2010). We note that the statistical challenges associated with the sacred areas analysis increases uncertainty of the estimates and our ability to define sweeping conclusions.

Displacing forest harvest from logging ban areas, which comprise the majority of the landscape, to sacred areas, which comprise a relatively small proportion of the landscape, may or may not have positive long-term impacts. Likely there will be trade-offs in shifting forest conservation from community-based institutions to centrally-regulated policies. Sacred area networks are important components of biodiversity conservation throughout Himalaya. Sacred areas are scattered throughout the landscape and protect a wide range of ecological niches and taxa, at multiple spatial scales, and even in surroundings with intensive human use (Bhagwat et al., 2005; Brandt et al., 2013; Dudley et al., 2010). Sacred areas have been resilient through centuries of social and political change, but are degrading now throughout the Himalaya, indicating that they are not resilient to current global change processes (Dudley et al., 2009; Verschuren et al., 2010). The long-term consequences of the degradation of such a broad-scale, resilient network of biodiversity protection is unknown. In regards to the logging ban, in the present, Chinese policy favors strong restrictions on timber harvest in Northwest Yunnan. However, the longevity, sustainability, or long-term effectiveness of a policy as restrictive as the logging ban is unknown.

Our results demonstrate that a combination of matching and panel data methods addressed many of the difficulties in estimating the effectiveness of different forms of forest protection in a spatially and temporally dynamic system. In particular, pre-estimation matching helped to balance control variables before the panel modeling, but the matching analysis alone was ill-equipped to deal with unobservable bias and with the assessment of change over multiple time-periods. The use of panel modeling permitted us to identify changes in effectiveness across time and space. Specifically, combining matching estimators and fixed effects difference-in-differences panel models enabled us to correct for both observed and unobserved sources of selection bias.

## 5. Conclusions

We analyzed the effectiveness of protected areas, a logging ban, and sacred areas in Northwest Yunnan, China. The overall impact

of forest protection policies was heterogeneous and depended on the form of protection, the type of forest protected, and the timing of the protection. We found two key results that have important implications for forest conservation policy and effectiveness assessment methodology. First, our results suggested that relying simply on gross forest cover to measure conservation effectiveness in areas with diverse forests can be misleading. In our study area, examining forest cover alone would have masked the distinct impacts that different protection strategies had on old growth and pine forests.

Second, a wide range of protection policy instruments worldwide, including PES (payment for ecosystem services) programs, logging bans, and community-managed forests, are increasingly implemented to complement protected area networks. Our research suggests that, in our setting at least, there is little benefit to overlapping policies. To the contrary, evidence in our study area suggest that as the logging ban was implemented, there was an increase in old growth forest harvest from sacred areas and pine forest harvest in protected areas. Thus, the implementation of one official policy may impact the effectiveness of community managed institutions and other official policies alike. In summary, our results highlight that to craft policies that can achieve multiple benefits of biodiversity conservation, livelihoods, and ecosystem services, there is an urgent need to include diverse forest types in policy impact assessments, and to consider how new policies interact with existing institutions.

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## Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.biocon.2014.09.043>.

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