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Article *in* Land Degradation and Development · April 2018

DOI: 10.1002/ldr.2970

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Received: 22 March 2017 | Revised: 7 April 2018 | Accepted: 11 April 2018
DOI: 10.1002/ldr.2970

RESEARCH ARTICLE

WILEY

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Funding information
Key Research Program of Frontier Sciences of the Chinese Academy of Sciences, Grant/Award Number: QYZDY-SSW-SMC014; Green Rubber project, Grant/Award Number: CRP FTA; Federal Ministry for Economic Cooperation and Development, Germany, Grant/Award Number: 13.1432.4-001.00; Yunnan provincial postdoctoral grant and Chinese Academy of Sciences President International Fellowship Initiative (CAS-PIFI), Grant/Award Number: 2017PC0035; China postdoc foundation, Grant/Award Number: 2017M613021

Abstract

Protected areas (PAs) are supposedly key refuges for the world's remaining biodiversity. Our study site, Xishuangbanna, harbors a high proportion of China's biodiversity but is threatened by rapid deforestation and expansion of monoculture rubber. We quantified the success of Xishuangbanna's PAs in preventing deforestation. Most previous analyses of PA effectiveness have insufficiently accounted for biases arising from PA location and establishment, because they overlooked the importance of site-matching in accounting for landscape change. We used matching methods to minimize such biases in comparing land use conversion rates inside and outside-PAs. By 2010, Xishuangbanna had 3,455.5 km^2 (~18%) designated as PAs. However, rubber occupied 22% of its land area and was expanding at a rate of 153.4 km^2/year. Between 1988 and 2010, conventional analysis showed a deforestation rate of 9.3 km^2/year. However, matching analysis showed a significantly higher rate of deforestation, 10.7 km^2/year, which resulted in the deforestation of ~11% of PA's land. We argue that PAs were less effective than had previously been thought. The situation worsened from 2002 to 2010, when the deforestation rate within PAs was actually higher than that of outside PAs, although this difference was not significant. The designated higher levels of protection in 'core' zones were also unsuccessful in preventing deforestation. At current rates, within the next 50 years, a further 16% of PAs would be deforested in Xishuangbanna. This could even be an underestimate, as without intervention, drivers of deforestation tend to accelerate. Therefore, reviewing and strengthening current PA management policies is essential.

KEYWORDS

deforestation, land use land cover, low land rainforest, protected areas, rubber monoculture, Xishuangbanna

1 | INTRODUCTION

Current rural development policies in China focus on harnessing technological innovation and promoting industrial plantations to intensify food and commodity production. Such policies can have negative environmental consequences, including loss of biodiversity. One possible mitigating strategy is to create and expand protected areas (PAs) (Jim & Xu, 2004); this is considered to be a key strategy for protecting biodiversity (Gaston, Jackson, Cantú-Salazar, & Cruz-Pi ón, 2008) especially in the biodiversity-rich tropics (Xu, Zhang, Liu, & McGowan, 2012). For example, establishment of PAs is one indicator included in the Millennium Development Goals for environmental sustainability (Chape, Harrison, Spalding, & Lysenko, 2005). Expansion (i.e., creation of new and or enlargement) of PAs is considered to be a successful outcome of conservation initiatives (Ervin, 2003). Approximately 23% of the tropical forest biome is now under some form of protection (UNEP-WCMC, 2008). Outside PAs (hereafter called non-PAs), loss and degradation of natural habitats are continuing at accelerated rates. Thus, it is essential to assess how far PAs are truly succeeding in halting such destruction (Cuenca, Arriagada, & Echeverría, 2016; Gaveau et al., 2009; Green et al., 2013; Nagendra, 2008). In this study, we analyzed a data set from China that had been recorded over two decades (from 1988 to 2010) of monitoring, in order to test how effective PAs have been in preventing deforestation and the expansion of industrial plantations.

1.1 | Xishuangbanna and rubber in China

Xishuangbanna in Southwest China is a part of the globally recognized Indo-Burma biodiversity hotspot (Myers, Mittermeier, Mittermeier, da Fonseca, & Kent, 2000) recording a disproportionate fraction of China's biodiversity, supporting ~25% of its fauna and flora on only 0.2% of its landmass (Liu et al., 2001). Recently, in the tropics, rubber plantations (*Hevea brasiliensis*) have expanded rapidly (Dewi et al., 2017) and now occupy 2.1 million ha across Southern China (Li & Fox, 2012), including Xishuangbanna (Li, Aide, Ma, Liu, & Cao, 2007). Expansion has been driven by the introduction of policies facilitating the replacement of traditional shifting cultivation by modern plantations (Fox, Castella, Ziegler, & Westley, 2014). Nevertheless, China's rubber production is still inadequate to meet its national demand. China currently uses 30% of global rubber production (FAO, 2010). Thus, the economic incentive to expand rubber is likely to continue. If conversion of land to rubber continues in Xishuangbanna, a high proportion of China's biodiversity is at risk.

1.2 | Current status of PAs in China and Xishuangbanna

PAs are at the heart of China's conservation policy (Fangliang, 2009; Xu et al., 2008) although we acknowledge a disproportionate distribution of PAs network within China at the moment (Xu et al., 2017). The first PA (Dinghu Shan) was established in 1956 in Guangdong province (Jim & Xu, 2004). By 2009, China had 2,541 PAs, accounting for 15% of its total land area (Xu et al., 2012). The administrative level (national, provincial or county) at which PAs are managed depends on their ecological value and the degree of human disturbance. However, PAs are often poorly managed due to lack of staff, funds, proper management agencies, and through conflicts between governing bodies (Jim & Xu, 2004). Some studies (Nagendra, 2008) have recorded rates of habitat loss inside PAs were higher than before the reserve was created. The flagship Wolong NR for panda conservation is one such example from China (Liu et al., 2001).

In accordance with China's commitment to expanding a national PA network, two PAs were established in Xishuangbanna in 1959 and 1993 (~241,776 ha) (Chen, Yi, et al., 2016). In 1991, Nabanhe PA was established (~26,600 ha) with Bulong PA following in 2009 (~36,000 ha) (Chongrui, 2009). To date, 18% (~346,200 ha) of Xishuangbanna lies within PAs (Li et al., 2007).

1.3 | Significance of PAs in Xishuangbanna

Within China, many species, including gaurs (*Bos frontalis*), leopards (*Panthera pardus fusca*), primates (IUCN, 1993; Li et al., 2007), and Asian elephants (*Elephas maximus*; Chen, Marino, et al., 2016; Zhang, 2011), are confined to lowland rainforest in Xishuangbanna, a habitat that has been mostly converted to rubber. Due to habitat loss, the Asian elephant is now restricted to only three PAs (Lin et al., 2008). Today, little natural forest exists in non-PAs in Xishuangbanna, particularly in the lowlands, whereas rubber has expanded from 264 to 23,616 ha during the past 30 years (Chen, Yi, et al., 2016) even within reserves covering a large area.

1.4 | Assessing the effectiveness of PAs

Although the number and the extent of PAs in a country could be considered to represent the political commitment to biodiversity conservation (Chape et al., 2005), establishment of PAs is a hollow achievement unless they truly meet conservation targets. Previous studies in several countries have assessed the effectiveness of PAs in their specific contexts (Andam, Ferraro, Pfaff, Sanchez-Azofeifa, & Robalino, 2008; Bruner, Gullison, Rice, & Fonseca, 2001; Mascia et al., 2014) including in China (Brandt, Butsic, Schwab, Kuemmerle, & Radeloff, 2015; Ren et al., 2015; Xu & Melick, 2007). These studies mainly compared deforestation rates in PAs with non-PAs (Defries, Hansen, Newton, & Hansen, 2005; Nagendra, 2008; Soares et al., 2010). In principle, PAs may be considered effective if their deforestation rates are lower than non-PAs (Joppa & Pfaff, 2009), although, in reality, this sets a very low bar. However, few studies have controlled for other factors that might determine deforestation and biodiversity loss, such as topography and the distance of PAs from markets. Thus, such studies as Bruner et al. (2001) could have overestimated the role of protection, as PAs are typically established on infertile soils or in inaccessible locations with lower pressures for land conversion (Venter et al., 2014). In some studies that have controlled for location-related bias, PAs have been shown to be effective in reducing deforestation (Joppa & Pfaff, 2011; Nelson & Chomitz, 2011; Soares et al., 2010) but, in other studies, they have been ineffective (Liu et al., 2001). Out of 118 studies reviewed, Geldmann et al. (2013) concluded that PAs have improved the conservation of forest habitats but that it could not be determined whether this has translated into effective species conservation. To improve the conservation benefits of PAs, Montesino Pouzols et al. (2014) stressed the need to repeat quantitative assessments of the effectiveness of PAs at multiple scales (globally, regionally, and nationally), with the goal of reducing further biodiversity loss within the worldwide PAs network.

Several previous studies in Xishuangbanna have examined threats to biodiversity (Lin et al., 2008), including rubber expansion (Chen, Yi, et al., 2016), but none have controlled for location bias and few have systematically investigated the drivers of degradation (Ren et al., 2015). In this study, we used matching techniques to control for the location-related bias of PAs establishment (Andam et al., 2008; Gaveau et al., 2009; Pfaff, Robalino, Sanchez-azofeifa, Andam, & Ferraro, 2009) to test whether the establishment of PAs provides any intrinsic protection.

If authorities were to randomly select the land assigned for PAs, then it would be straightforward to estimate the impact of protection by just considering a conventional analysis of land use conversion through time both in PAs and non-PAs, but in reality, across the world, almost all PAs are established in locations that are considered of particular importance for conservation, often on land that has the least commercial value and has been proven to be relatively inaccessible to humans. So the interior and surroundings of PAs are likely to differ with respect to many environmental and economic variables (Dewi, van Noordwijk, Ekadinata, & Pfund, 2013; Mas, 2005). To control for these differences, we applied matching methods, to address the questions:

a. Is habitat conversion lower inside PAs than in comparable unprotected areas?
b. Does the effectiveness of PAs differ among different management zones within PAs?

Assessment of PA's effectiveness using propensity score matching (PSM) techniques has been conducted in some countries (Andam, Ferraro, Sims, Healy, & Holland, 2010; Blackman, Pfaff, & Robalino, 2015; Gaveau et al., 2009; Pfaff et al., 2009) but not in China.

2 | MATERIALS AND METHODS

2.1 | Study site

Xishuangbanna (19,164 km^2) is located in Southwest China. It is mountainous (475–2,428 m; Li, Ma, Aide, & Liu, 2008), with distinct natural forest types including tropical seasonal rain forest (<900 m), tropical montane rainforest (700–1,500 m), monsoon forest (<900 m), monsoon forest located on limestone (<800 m), subtropical evergreen broad-leaved forest (1,000–1,500 m), and montane evergreen broad-leaved forest (1,000–2,400 m; Zhang & Cao, 1995; Figure 1).

There are two legal types of land tenure in Xishuangbanna: state owned land (PAs) and communal land (agriculture and forestry). Yunnan Provincial Forestry Bureau is responsible for the management of Xishuangbanna Nature Reserves (XNRs). They are under the overarching jurisdiction of the Ministry of Forestry (IUCN, 1993). XNRs are

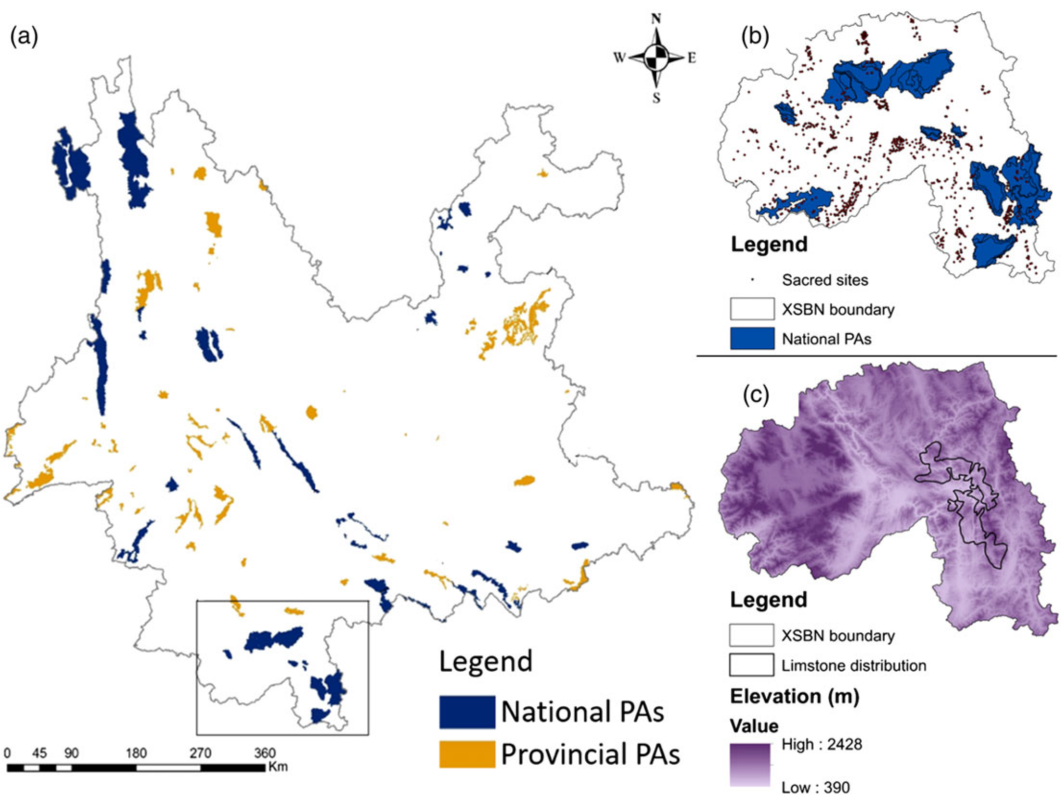

FIGURE 1 (a) Spatial distribution of protected areas (PAs) of Yunnan province in China (Zomer, Xu, Wang, Trabucco, & Li, 2015). The box indicates the location of Xishuangbanna, (b) the distribution of PAs in Xishuangbanna, to account for county level protection in locally managed sacred lands that were considered to be PAs, and (c) elevation map of Xishuangbanna showing geographical location of limestone forest [Colour figure can be viewed at wileyonlinelibrary.com]

divided into three zones; core, buffer, and experimental land for management purposes. The core zone is designated for strict protection, whereas the buffer zone serves to enhance the protection of core zone. Limited agriculture and subsistence hunting are permitted in the experimental zone (Nepal, 2002). XNRs have been part of the 'UNESCO Man and Biosphere Programme' since 1993 and are assigned to IUCN Management Category IV (IUCN, 1993).

2.2 | Site selection of PAs in Xishuangbanna

For our analysis, we considered two sets of PAs; IUCN recognized PAs with their subsections (Bertzky et al., 2012) and sacred lands which are community managed PAs. Although Clarke (2002) and Wen (1997) mentioned that Xishuangbanna hosts categories of PAs as NRs, forest parks, county level NRs further reliable information on these types of PAs were unavailable. Official information on the boundaries of county level NRs are difficult to obtain, but NRs almost always coincide with sacred lands. There are no provincial level NRs in Xishuangbanna, because as it was government policy to apply for national reserve status to promote the tourism industry, which is extremely important to Xishuangbanna.

2.3 | Mapping land use changes and other variables affecting PAs establishment

The most important factors determining the spatial patterns of land use, deforestation, and effective conservation are likely to be slope, elevation, distance to roads, and indicators of agricultural and logging potential, such as soil and forest types (Andam et al., 2008; Gaveau et al., 2009; Kaimowitz & Angelsen, 1998; Mas, 2005). For our study, we tested the impact of elevation, slope, soil type (limestone *vs.* non-limestone), distance to roads, and land use types. We did not investigate historical land use changes. Most rubber expansion has been at the expense of forests; therefore, deforestation and rubber expansion were both treated as dependent variables.

Administrative data, including boundaries of NRs, digital elevation model (DEM; scale 1:50,000, 25 m resolution) were obtained from the Centre for Mountain Ecosystem Studies, Kunming Institute of Botany, Chinese Academy of Sciences (Chen, Yi, et al., 2016). Elevation and slope layers were derived from the DEM. We created limestone distribution maps referring Zhu, Wang, and Baogui (1998) and overlaid it on the 30 * 30 m grid layer. Based on a Landsat TM image and a Landsat ETM+ image (path/row number: 130/45), rubber was mapped. A recently published map by Xu, Grumbine, and Beckschäfer (2014) was used for 2010 and the corresponding RapidEye satellite images were reanalyzed, using a refined classification scheme that enabled rubber and other land use types to be distinguished. Spatial dataset created by processing and quantifying land use types was empirically quantified in Google Earth Quickbird imagery using membership function and nearest neighbor classifiers in eCognition 8.0 (Trimble, USA). The membership function classifier was applied for identification of different land uses in Landsat images. Freely accessible DigitalGlobe archives were used to assess the accuracy of the classifications (Chen, Yi, et al., 2016).

Raster layers of land uses (1988, 2002, and 2010) were overlaid with elevation, slope, soil layers, administrative boundaries, and the boundaries of PAs. For each land use type, mean elevation, slope, soil, and distance to roads were calculated within and outside PAs. We identified similar featured land cells referring to elevation, slope, soil, and distance to roads in PAs and non-PAs to compare land conversion rates treating the data as a conversion probability from each timepoint to the next. We conducted two levels of analysis: (a) All IUCN recognized PAs (n = 7) established before 2010 plus all the locally protected sacred forest and (b) IUCN recognized PAs only. We treated PA and non-PA status as a binary variable and defined protected cells as treatment and unprotected cells as control.

Using Equation (1), annual land use conversion rates were calculated assuming linear conversions both in PAs and non-PAs as well as in different PAs zones (core, buffer, and transition zones).

$$\text{LU change} = \frac{(A_{t2} - A_{t1})}{(t_2 - t_1)}, \tag{1}$$

where

LU change = land use change rate per year (km^2 yr^{-1}),

t_2 and t_1 = time periods (years)

A_{t_2} and A_{t_2} = areas of given land use at times t_2 and t_1 (km^2)

To account for temporal variation, we analyzed the data from 1988 to 2002 and 2002 to 2010 separately.

2.4 | Data analysis

2.4.1 | Controlling for geographic bias in PAs locations using matching methods

We used PSM, to reduce the potential bias emerging from nonrandom geographic placement of PAs. By matching, data were preprocessed prior to the parametric analysis to consider the probability of a land cell being protected. A probit model was used to generate this probability of protection giving more weight to the variables which are crucial in determining protection and deforestation (Pfaff, Robalino, Lima, Sandoval, & Herrera, 2014).

First, we estimated the PS, which is defined as the conditional probability (between 0 and 1) of assigning a unit (here a grid cell) to a particular treatment condition (i.e., the likelihood of receiving protection), given a set of observed covariates (Olmos & Govindasamy, 2015). PS can be calculated using two methods: (a) logistic regression and (b) classification and regression tree analysis (Thavaneswaran & Lix, 2008). We used logistic regression which consists of computing the probability of an event to occur under certain conditions.

It first estimated the PS (e [*xi*]),

where

$$e(x_i) = \Pr(z_i = 1 | x_i), \tag{2}$$

$e(xi)$ = estimated propensity score

z = treatment

i = treatment condition

i = 1 (protection)

i = 0 (no protection)

x_i = observed value of variables (elevation, slope, distance to road, soil).

Then, the logistic regression for covariates were calculated as follows:

$$\ln\frac{e(x_i)}{1-e(x_i)} = \ln\frac{\Pr(z_i = 1|x_i)}{1-\Pr(z_i = 1|x_i)}. \quad (3)$$

And for all covariate, the regression equation is

$$e\,(Wi) = b_0 + b_1W_1 + b_2W_2 + b_3W_3 + \ldots + b_iW_i, \quad (4)$$

where

b_0 = the intercept,

bi = the regression coefficient of ith covariate,

Wi = the treatment variables and covariates.

Using the *matchit* function in R, we estimated PS, we started PSM by pairing up treatment and control units which had the most similar PS. Thus, PSM evaluates what would have happened had a treatment (protection) not been applied (Andam et al., 2008; Ferraro & Pattanayak, 2006; Rosenbaum & Rubin, 1985). Then, we used generalized linear model (*glm*) function to check whether all variables incorporated in the model were significant. Significant variables were kept in the model. We employed the function *match.data* to generate data sets that have matched cases for the follow-up analysis. Matching methods available in MatchIt package in R environment 3.3.1 (R core Team, 2016) were used (Carranza, Balmford, Kapos, & Manica, 2014).

We selected the nearest neighbor matching method, which is still valid when there are more controls (non-PA, n = 279,382) than treatments (PA, n = 28,603), as it minimizes the absolute difference between the estimated PS for the control (PS_j) and treatment groups (PS_i). In this method, control and treatment subjects are randomly ordered. Then the first treated subject is selected along with the control subject with the nearest PS.

$$C\,(PS_i) = \min_j \left\| PS_i - PS_j \right\| C(\mathrm{Pi}) = \min\vee\mathrm{Pi}-\mathrm{Pj}, \quad (5)$$

where

C(PSi) = the group of control subjects *j* matched to treated subjects *i* (on the estimated PS)

PSi = the estimated propensity score for the treated subjects *i*

PSj = the estimated propensity score for the control subjects *j*

To compare the matched sets for each land use category in each time period (1988–2002 and 2002–2010), we used one-tailed paired-sample *t* test where the response variable was deforestation or rubber expansion.

3 | RESULTS

3.1 | Covariate analysis

Covariate analysis tests how geographic factors affect existing land use practices and deforestation (Andam et al., 2008). In 2010, based on mean values, protected forest plots were closer to the roads (7.6 ± 3.4 m) than their unprotected counterparts (8.1 ± 5.4 m), but the reverse was true for rubber plantations (PAs 7.7 ± 5.8 m, non-PAs 6.8 ± 4.2 m; Table S1). The mean elevation of rubber in PAs was lower than non-PAs. Mean elevation and mean slope of forest and rubber in PAs and non-PAs were not significantly different (Figure 2).

3.2 | Deforestation and associated land cover changes

In 1988, natural forest comprised ~79% (15,129 km^2) of sample cells but, in 2010, this was reduced by 23.4%. Monoculture rubber was

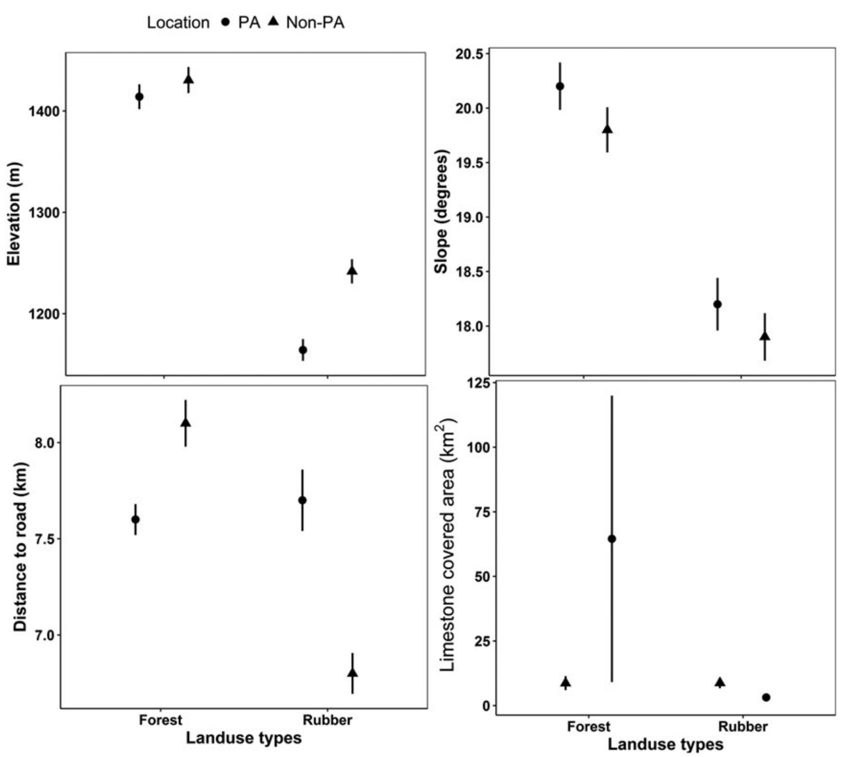

FIGURE 2 Distribution of covariates (slope, elevation, soil and distance to road; mean ± SE) for forest and rubber land in protected areas (PAs) and non-PAs in Xishuangbanna

responsible for 75% of this deforestation, followed by agriculture 20.8% (Table S2). This rapid expansion of rubber has replaced the diverse tropical seasonal rainforests that previously occupied the most productive areas for rubber (Warren-Thomas, Dolman, & Edwards, 2015).

3.3 | PSM analysis–Land use changes of PAs and non-PAs

In the matched data set for 1988–2002, deforestation of PAs was significantly lower than that of non-PAs (−1.51% < −2.76%). In contrast, during 2002–2010, deforestation in PAs was higher than non–PAs (−6.09% > −5.24%; Table 1) but it was not significant (1988–2002, t = −1.34, df = 731, 95% CI [∞, 3.94], p = .045), (2002–2010, t = 2.41, df = 224, 95% CI [∞, 16.192], p = .49 ; Table S3).

During both periods, rubber expansion percentages were lower in PAs compared with non-PAs (0.95% < 8.3% and 6.7% <14.9%) but during 2002–2010, this value was not significant whereas for 1988–2002, this was significant (t = −1.94, df = 260, 95% CI [∞, −0.0132519], p = .014) indicating the effectiveness of PAs at the start.

3.4 | Effectiveness of PAs

During 1988–2002 and 2002–2010, deforestation rates (after matching) in PAs (3.3 and 23.6 km^2/year) were lower than non-PAs (31.6 and 104.9 km^2/year), this deforestation rate was significantly lower during 1988–2002 (Table S3) after controlling for location bias.

For 1988–2002, rubber expansion rate (after matching) in-PAs (2.1 km^2/year) was significantly lower than in non-PAs (94.9 km^2/year; Table S3). During 2002–2010, rubber expansion rate in-PAs (26.3 km^2/year) was lower than non-PAs (300 km^2/year; Table S4).

According to the conventional analysis of national PAs, during 1988–2002, the deforestation rates of buffer (1.41 km^2/year) < core (2.2 km^2/year) < transition (2.3 km^2/year) whereas during 2002–2010, these rates were core (1.1 km^2/year) < transition (3.6 km^2/year) < buffer (3.8 km^2/year; Table S5).

One of the important observation for rubber during 1988–2002 was that it showed 0 expansion in core zone; transition (1.2 km^2/year) > buffer (0.3 km^2/year) but from 2002 to 2010 period, rubber has invaded core zones of national PAs (0.9 km^2/year) followed by buffer (3.6 km^2/year) and by an exponential increase rate in transition zone (22.3 km^2/year) (Table 2).

In contrast to this positive rubber expansion during 2002–2010, deforestation of core zones reduced by 1.56% compared with 1988–2002. This might be due to the reduction of other main land uses such as tea and farmland (Table S5). However, during 2002–

TABLE 2 Land use change percentages (± SE) of buffer, core and transition zones of PAs for 1988–2002 and 2002–2010

Land use	Zone type	Land use change (%) 1988–2002	2002–2010
Farmland	Buffer	−0.01 ± 0.0005	−0.58 ± 0.2000
	Core	0.02 ± 0.0003	0.13 ± 0.0100
	Transition	3.59 ± 0.0020	0.33 ± 0.3000
Forest	Buffer	−2.57 ± 0.3700	−3.95 ± 0.4100
	Core	−2.20 ± 0.5900	−0.64 ± 0.0400
	Transition	−2.49 ± 0.1600	−2.24 ± 0.2900
Shrub	Buffer	−0.08 ± 0.0300	−0.52 ± 0.0200
	Core	−0.19 ± 0.0200	0.28 ± 0.3000
	Transition	−0.31 ± 0.0200	−0.29 ± 0.1200
Tea	Buffer	−0.02 ± 0.0010	−0.50 ± 0.2300
	Core	0.01 ± 0.0050	−0.05 ± 0.0005
	Transition	−0.11 ± 0.0010	−2.36 ± 0.0600
Rubber	Buffer	0.54 ± 0.1100	3.71 ± 0.3700
	Core	0.00 ± 0.0000	0.50 ± 0.1000
	Transition	1.34 ± 0.4000	13.83 ± 0.7500
Construction	Buffer	0.00 ± 0.0000	0.04 ± 0.0010
	Core	0.00 ± 0.0000	0.01 ± 0.0003
	Transition	0.00 ± 0.0000	0.45 ± 0.0200
Banana	Buffer	0.00 ± 0.0000	0.10 ± 0.0100
	Core	0.00 ± 0.0000	0.001 ± 0.0000
	Transition	0.00 ± 0.0000	0.42 ± 0.3200
Roads	Buffer	0.00 ± 0.0000	0.00 ± 0.0000
	Core	0.00 ± 0.0000	0.00 ± 0.0000
	Transition	0.00 ± 0.0000	0.06 ± 0.0100
Other	Buffer	−0.13 ± 0.0010	0.04 ± 0.0030
	Core	0.00 ± 0.0000	0.00 ± 0.0000
	Transition	−0.06 ± 0.0020	−0.53 ± 0.0800

TABLE 1 Summary of land use change percentages (± SE) in protected areas (PAs) and non-PAs after and before propensity score matching for 1988–2002 and 2002–2010

	Before matching				After matching			
	PA		Non-PA		PA		Non-PA	
Land use	1988–2002	2002–2010	1988–2002	2002–2010	1988–2002	2002–2010	1988–2002	2002–2010
Banana	0.00 ± 0.00	0.20 ± 0.05	0.00 ± 0.00	1.10 ± 0.51	0.00 ± 0.00	0.20 ± 0.01	0.00 ± 0.00	0.03 ± 0.02
Construction	0.00 ± 0.00	0.20 ± 0.01	0.00 ± 0.00	1.26 ± 0.02	0.00 ± 0.00	0.20 ± 0.001	0.00 ± 0.00	0.02 ± 0.01
Farmland	−0.16 ± 0.14	0.27 ± 0.21	2.35 ± 0.003	0.77 ± 0.003	2.69 ± 0.36	0.21 ± 0.004	0.69 ± 0.08	0.35 ± 0.21
Forest	−1.07 ± 0.69	−5.60 ± 0.18	−5.27 ± 0.44	−17.6 ± 0.52	−1.51*± 0.03	−6.09 ± 0.79	−2.76 ± 0.75	−5.24 ± 2.38
Other	−1.20 ± 0.03	−0.02 ± 0.02	0.26 ± 0.05	−0.21 ± 0.05	0.00 ± 0.00	−0.02 ± 0.01	0.26*± 0.03	−0.21 ± 0.03
Road	0.00 ± 0.00	0.02 ± 0.01	0.00 ± 0.00	0.05 ± 0.03	0.00 ± 0.00	0.04 ± 0.002	0.00 ± 0.00	0.05 ± 0.001
Rubber	0.80 ± 0.11	5.67 ± 0.8	7.43 ± 0.002	12.4 ± 0.02	0.95*± 0.03	6.79 ± 0.04	8.27*± 0.22	15.0 ± 0.17
Shrub	−20.55 ± 0.4	−0.15 ± 0.4	4.15 ± 0.01	−0.14 ± 0.01	−34.4*± 0.54	−0.12*± 0.01	−3.21*± 0.09	−0.06*± 0.01
Tea	0.09 ± 0.48	−1.00 ± 0.35	−0.20 ± 0.06	−1.51 ± 0.04	0.44*± 0.02	−1.00 ± 0.02	0.02 ± 0.03	0.13 ± 0.26
Water	−0.09 ± 0.05	0.31 ± 0.08	0.01 ± 0.05	−0.06 ± 0.05	0.49 ± 0.06	−0.18 ± 0.02	0.01 ± 0.01	0.00 ± 0.01

*$p \leq .05$, **$p \leq .001$ (Table S3).

2010, banana plantations expanded (transition [0.7 km^2/year] > buffer [0.1 km^2/year] > core [0.001 km^2/year]).

Map analysis showed that rubber has started spreading into the highly protected core zones, despite their protected status; both invading forest land and also expanding to higher altitudes (from 0 m to ~2,100 m in Nabanhe and Bulong; Figure 3).

4 | DISCUSSION

4.1 | Propensity score matching

Our study confirms overestimation of PA effectiveness when conventional methods (not controlling for PAs location bias) are applied (Tables S6 and S7). Using PSM, which takes this bias into account (Andam et al., 2008), we were able to demonstrate that PAs in Xishuangbanna showed a weakening in their effectiveness through time with respect to preventing deforestation and rubber expansion (Table 1). Some studies have shown that PAs are effective in reducing land conversions inside them but not in their surroundings (Dewi et al., 2013). Matching methods in our analysis did not account for this leakage and thus may have still overestimated PAs effectiveness.

Most PAs in Xishuangbanna are located at higher elevations, between 500 and 2,400 m, and on steep slopes where rubber growth is suboptimal. However, rubber continues to be planted at higher elevations in PAs, such as in Nabanhe, despite their protection status and suboptimal growing conditions. According to Yi, Cannon, Chen, Ye, and Swetnam (2014), rubber at higher elevations takes longer to reach maturity and produces lower yields. High-elevation rubber expansion causes natural habitat degradation that is of low economic benefit.

4.2 | Land use changes and deforestation drivers in Xishuangbanna

Deforestation is, by definition, a conversion of forest to non-forest. Most rubber expansion has been at the expense of previously forested areas (Qiu, 2009; Xu et al., 2014). In the context of Xishuangbanna, we did not include plantations under forestry (FAO, 2010). Although overall deforestation and rubber expansion rates are higher in non-PAs than in PAs, once location biases were accounted for we showed that

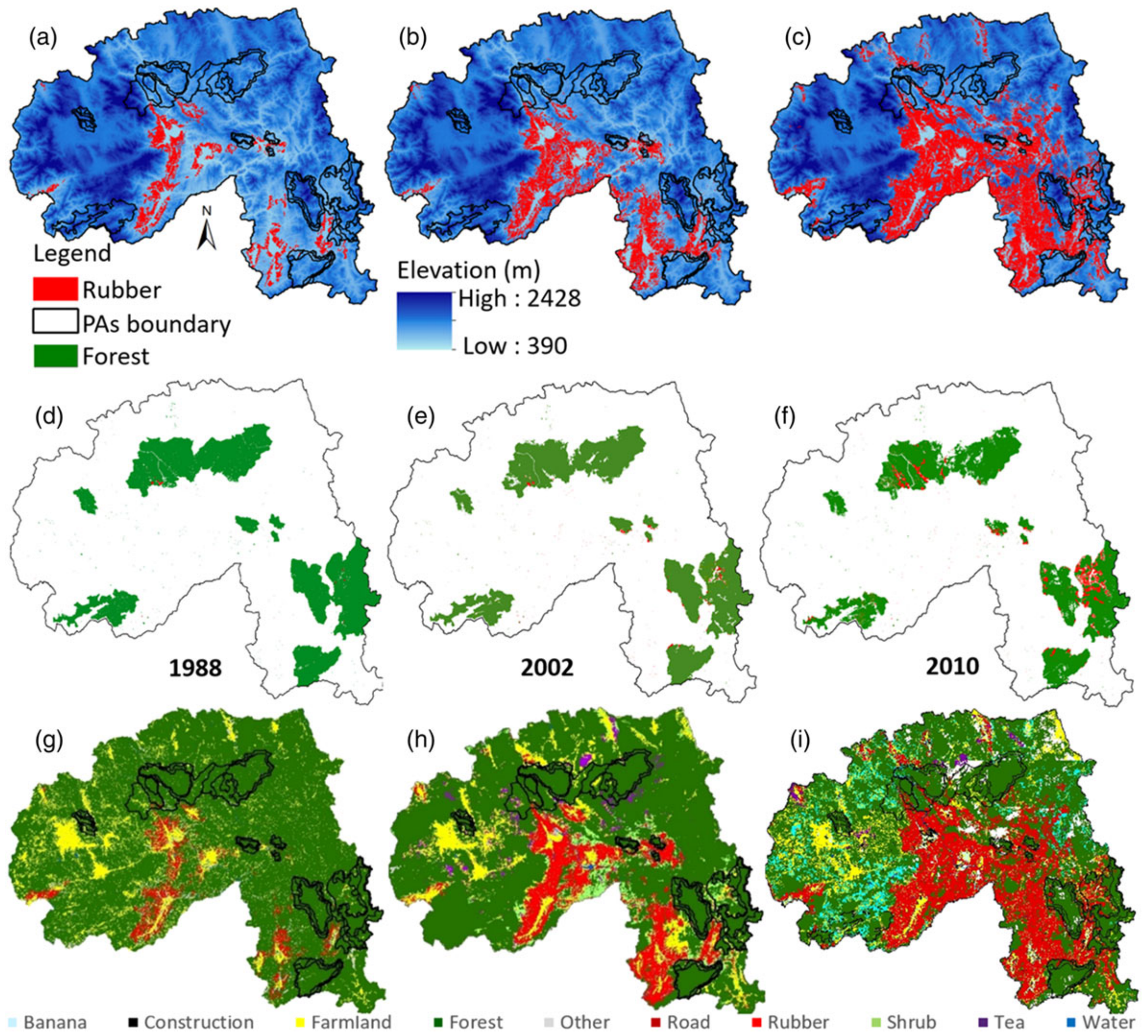

FIGURE 3 Rubber distribution in protected areas (PAs) and non-PAs with respect to the elevation in Xishuangbanna for the periods of (a) 1988, (b) 2002, and (c) 2010. Rubber expansion into PAs in (d) 1988, (e) 2002, and (f) 2010. Total land use land cover map with all deforestation within (g) 1988, (h) 2002, and (i) 2010 [Colour figure can be viewed at wileyonlinelibrary.com]

PAs in Xishuangbanna were not effective in preventing land conversion (Table S8 and S9). Our findings support the suggestions of earlier studies that pressures outside reserves may cause long-term conservation failures within PAs (Porter-Bolland et al., 2012). Our analysis draws attention to the ongoing failures of conservation strategies in China, where PAs are often inadequately managed (Xu et al., 2012; Xu & Melick, 2007). Focusing conservation efforts on PAs has drawn attention away from the other pillars of conservation, such as endangered species protection which is also failing in China and across SE Asia, where PAs are often inadequately managed (Harrison et al., 2016). One of the most significant outcomes of our analysis was that although PAs initially seemed to be somewhat effective in preventing deforestation (Table S1), once the land available for rubber became limited, PAs served as a source of land for further rubber expansion. The 2002–2010 analysis showed that after controlling for location biases, deforestation was actually higher inside PAs than outside (Table S8).

The historical change of land ownership in China further evidences these changes. From the early 1950s, the government established large-scale state-owned rubber farms in Xishuangbanna, which were primarily responsible for the pre-1980s rubber boom. However, with the introduction of the 'Household Responsibility System' in early 1980s, smallholder rubber farmers were empowered to plant rubber (Chapman, 1991). Thus, monoculture rubber started replacing Xishuangbanna lowland forest in 1960's and swidden agriculture in 1990's. This situation was exacerbated by inadequate policy definitions, as government authorities did not distinguish between rubber and natural forest in their national statistics (van Noordwijk, Tata, Xu, Dewi, & Minang, 2012). After 1996, the government deregulated and decentralized PA administration in China (Jim & Xu, 2004), which may also have contributed to the accelerated degradation of natural habitats within Xishuangbanna's PAs over the study period.

In 2000, the national government implemented the 'Western Development Strategy' to combat poverty and to industrialize western provinces, including Yunnan. In 2001, China entered the World Trade Organization proposed formation of the China-ASEAN Free Trade Area. These political changes were catalysts for development-based economic growth. The government invested in infrastructure and other development-related activities, thus creating favorable conditions of institutional and economic power that attracted outside investors. This drove further rubber monoculture expansion in Xishuangbanna (Xu & Salas, 2003).

From 1988 to 2003, average rural incomes in Xishuangbanna increased tenfold (Stone, 2009), mainly as a result of rubber. Current rubber yield prices are around 15,000 RMB ha/year whereas farmers can only get 2,000–3,000 RMB ha/year from tea or rice cultivation (Qiu, 2009). In average, rubber latex price is 2.57 USD kg^{-1} in China whereas the global market price is 1.98 USD kg^{-1}. The rapid rate of rubber expansion in Xishuangbanna is therefore not surprising. Communities living within PAs or with land next to PAs may feel excluded from their neighbor's economic opportunities, even if land within PAs is suboptimal for rubber. However, these rewarding economic incentives are associated with undeniable negative environmental impacts (Ziegler et al. 2009).

In addition to accelerated rubber expansion during 2002–2010, we observed other important changes to land use practices (Table S10), including the introduction of short term cash-crops and infrastructure development (roads and construction) to facilitate the emerging economy based on plantations and industries. Banana, a fast-growing cash-crop, is replacing paddy rice cultivation in Xishuangbanna due to its profitability both in PAs and non-PAs. Existing forests are increasingly limited to higher elevation and steeper slopes. Topography, therefore, is perhaps the best explanation for an overall lower deforestation rate within PAs (i.e., before controlling for location biases) and the continued existence of forest in non-PAs.

4.3 | Effect of zonal management within PAs in preventing land conversion and deforestation

The zonal division of PAs was developed to give the greatest protection to the highest priority, core areas by allowing low impact extractive activities in buffer areas (Ebregt & Greve, 2000). However, in Xishuangbanna, rubber showed high positive expansion rates at the expense of forest, inside all zones of all the national level PAs (transition 15.1% > buffer 4.3% > core 0.5%) and also in sacred sites (Tables S10 and S11) indicating failures in management of the zonal divisions and questioning the implementation of both national level and locally managed PA policies.

In Xishuangbanna, rubber, banana, construction, and roads all increased in all the three PA zones representing a high risk for long-term PA management goals. Previous studies have already questioned whether stricter PAs are effective in achieving environmental objectives (Ferraro et al., 2013). In Xishuangbanna, even where buffer and transitional zones exist, the protection of core zones has been compromised. Less strict protection may be a tool to achieve workable compromises between livelihood and conservation goals, but only where management is effective.

5 | CONCLUSION

Xishuangbanna, one of the most biodiverse regions of China, is experiencing rapid deforestation and land cover changes due to monoculture rubber expansion and other recently introduced short-term cash crops, such as banana. Prior to our analysis to investigate efficiency of regional conservation strategy through PAs and temporal land use changes effects, we employed matching analysis to account for biases in the nonrandom implementation of PAs. Approximately, 16% of PA land was deforested between 1988 and 2010. Land conversion rates inside PAs were not significantly different to those of non-PAs, after controlling for location biases. In fact, deforestation rates and rubber expansion rates were actually higher inside PAs from 2002 to 2010. There were accelerated rates of rubber expansion during both periods. PA management has been ineffective in reducing deforestation and expansion of rubber. We emphasize the need to review and strengthen current PAs policies and governance in Xishuangbanna and to implement appropriate methods to monitor PA performance. A recent study by Zomer et al. (2015) also stressed the need to integrate policy with information on species distribution and the predicted effects of climate change. Xishuangbanna has experienced a recent widespread reduction of wildlife populations. Several species of critical conservation concern are now considered locally extinct (Sreekar, Zhang, Xu, & Harrison, 2015). Therefore, there is also an urgent need to integrate PA management with the protection of endangered species across the broader landscape (Harrison et al., 2016).

ACKNOWLEDGMENTS

We thank Yirga Alemu, Bi Yingfeng, Lu Yang, Mingxu Zhao, Dhanushka Nadeeshan, Geng Yanfei, Lili Li, Xia Xue, Kim Kwangchol, Wang Mingchen, Alice C. Hughes, and Fiona R. Worthy for their contribution to this study and four reviewers, including Meine van Noordwijk, for improving the manuscript. G. G. O. Dossa was supported by Yunnan provincial postdoctoral grant and CAS-PIFI grant 2017PC0035 with China postdoc foundation grant 2017 M613021. This study was funded by the Green Rubber project, CRP FTA, Chinese Academy of Sciences' Frontier Science Key Project (grant QYZDY-SSW-SMC014) and the Federal Ministry for Economic Cooperation and Development, Germany (grant 13.1432.7-001.00).

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

How to cite this article: Sarathchandra C, Dossa GGO, Ranjitkar NB, et al. Effectiveness of protected areas in preventing rubber expansion and deforestation in Xishuangbanna, Southwest China. *Land Degrad Dev*. 2018;1–11. https://doi.org/10.1002/ldr.2970