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Impact of land-use zoning for forest protection and production on forest cover changes in Bhutan

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ABSTRACT

Bhutan is characterized by a landscape dominated by forests. A substantial share of these forests is dedicated to nature conservation, with an extensive protected area network connected by biological corridors. Forestlands are also partly allocated to timber production, including forest management units subjected to strict regulations. We assessed the effectiveness of these various land-use zoning units to protect forest cover. We used a matching procedure to control for covariates and obtain robust estimates of the impact of each type of unit on forest cover changes during the 2000s. We also investigated subsets of the protected area network to test for effectiveness heterogeneities within this network. Our results showed that protected areas prevented 63% of the forest loss expected in forestlands under this protection status. These units also curtailed forest gain. Long-established protected areas were more effective at avoiding forest loss than recent ones, while the levels of stringency and operationality of protected areas had no differentiable impact on forest loss. We detected more forest loss in forests surrounding protected areas compared to more distant forestlands, showing a leakage effect. Biological corridors had no impact on forest loss and gain. Forest management units decreased forest loss by half. After accounting for the selection bias, this study demonstrated the effectiveness of land use zoning for forest conservation in Bhutan.

1. Introduction

According to the latest FAO Forest Resources Assessment, worldwide annual rates of net forest loss have more than halved between the 1990s and the 2010–2015 period (Keenan et al., 2015). Tropical deforestation also slowed, mostly due to decreasing deforestation rates in Brazil (Keenan et al., 2015). However, this reduction is contested by direct remote sensing observations, which measured a 62% increase in net humid tropical deforestation between the 1990s and the 2000s (Kim, Sexton, & Townshend, 2015). The tropics concentrated 32% of global forest loss in 2000–2012 (Heino et al., 2015). The fate of tropical forests thus remains of major concern, particularly in poor, tropical countries (Sloan & Sayer, 2015).

Although nonstate, market-driven governance regimes are yielding promising conservation outcomes (Heilmayr & Lambin, 2016), biodiversity conservation still largely depends on public interventions, including land use zoning (Lambin et al., 2014). Zoning consists of segmenting the landscape into units where human access and uses are legally restrained and limited to specific activities or agents according to their assignment, such as protection or production activities. The designation of natural areas under a protection status – i.e., protected areas – is a particular type of land-use zoning, commonly used for biodiversity protection (Andam, Ferraro, & Hanauer, 2013; Cuenca, Arriagada, & Echeverría, 2016; Geldmann et al., 2013; Hanauer & Canavire-Bacarreza, 2015; Joppa & Pfaff, 2010; Mascia et al., 2014; Miteva, Pattanayak, & Ferraro, 2012).

Globally, the share of the terrestrial realm designated as a protected area increased exponentially since the late 1950s and was estimated at 14.4% in 2014 (Ferraro & Pressey, 2015; Watson, Dudley, Segan, & Hockings, 2014). Areas under protection include 16.3% of the world forests and up to 26.6% of tropical forests (Morales-Hidalgo, Oswalt, & Somanathan, 2015), with great variability between countries and ecoregions (Schmitt et al., 2009; Watson et al., 2014). Downsizing, downgrading, or even degazettement of areas under protection is also

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Table 1
Protected areas and biological corridors of Bhutan.

Name	Type	Year of creation	Operational year (NSB, 2011)	IUCN category	Area (km^2)	Households (Wangchuk, 2007)	Settlement/km^2
Jigme Dorji	National Park	1974	1995	II	4324	1000	0.31
Bumdeling	Wildlife Sanctuary	1995	1998	IV	1537	136	0.15
Thrumshingla	National Park	1998	2000	II	908	1626	0.19
Toorsa[a]	Strict Nature Reserve	1993	/	Ia	611	na	0.03
Sakteng	Wildlife Sanctuary	1993	2003	IV	743	616	0.69
Jigme Singye Wangchuck	National Park	1995	1995	II	1727	950	0.65
Royal Manas	National Park	1966	1966	II	1024	650	0.57
Khaling[b]	Wildlife Sanctuary	1974	/	IV	338	na	0.42
Phipsoo	Wildlife Sanctuary	1993	/	IV	270	na	0.79
Wangchuck Centennial	National Park	2008	/	II	4922	na	0.13
North corridor[c]	Biological corridor	1999	/	VI	934	na	1.01
TSNR-JDNP[d]					149		0.33
JDNP-JSWNP[d]					275		1.00
TNP-BWS[d]					79		2.39
TNP-JSWNP-RMNP[d]					501		0.10
KWS-SWS[d]					160		0.01
JSWNP-RMNP-PWS[d]					376		0.85
RMNP-KWS[d]					212		0.99

Note: na: Not available.
[a] Recently renamed, in honor of the ruling King, as the Jigme Khesar Strict Nature Reserve.
[b] Recently renamed as Jomotshangkha Wildlife Sanctuary.
[c] The north corridor connects the Wangchuck Centennial Park with the four PAs in its surroundings.
[d] Name of BC is the abbreviation of the PAs it connects.

taking place (Mascia et al., 2014). Other forms of zoning, such as for extractive purposes, can also contribute to forest conservation (Bruggeman, Meyfroidt, & Lambin, 2015). Zoning also risks causing leakage by displacing land uses to the periphery of zones with restricted uses (Lambin & Meyfroidt, 2011).

Given variations in stringency and enforcement of land-use zoning policies, there is a need for empirical evidence on their effectiveness to support the design of future ecosystem conservation programs (Ferraro & Pressey, 2015; Gaveau, Linkie, SuyadiLevang, & Leader-Williams, 2009a; Heino et al., 2015; Miteva et al., 2012). Their ability to deliver desirable outcomes is evaluated in terms of both environmental and socio-economic impacts (Cuenca et al., 2016). The impact evaluation literature emphasizes that forest conservation outcomes of protected areas cannot rely on a simple comparison between rates of forest loss in protected and unprotected areas. Actually, selection of areas designated for protection is not random and potentially correlated with probability of forest loss. Protected areas tend to be located where opportunity costs of conversion to other land uses are low, such as areas that are remote, unpopulated, at high elevation, on steep slopes, or with reduced agricultural suitability. This partly explains their imperfect ecological representation (Watson et al., 2014). Accounting for this non-randomness of zoning is critical in assessing the causal impact of protection, i.e., to estimate avoided deforestation compared to deforestation that would have occurred in the absence of protection (Cuenca et al., 2016; Gaveau et al., 2009b).

The Kingdom of Bhutan is located in the Himalaya biodiversity hotspot, with a landscape dominated by forests (Bruggeman, Meyfroidt, & Lambin, 2016). The Bhutanese government has made environmental conservation a pillar of its development philosophy (Brooks, 2010; Jadin, Meyfroidt, & Lambin, 2015; Meyfroidt & Lambin, 2010). The designation of areas for nature protection has been promoted for several decades, with circa 43% of the country area (~38,000 km^2) and 33% of its forests being protected in 2010 (FAO, 2014; NSB, 2011). This extensive protected area network, connected by biological corridors, offers a great opportunity to test the effectiveness of these interventions. Furthermore, the Bhutanese forestry sector has been nationalized and is strictly regulated, with timber extraction confined to specific production units.

The objective of this study is to assess the impact of the zoning of forestlands, including protection and production units, on forest cover changes in Bhutan between 2001 and 02 and 2011. The study period follows the 1995 Forest and Nature Conservation Act, which guides forest management. The impact of zoning could depend on location and characteristics of zoning units, and on causes of forest cover changes (Ferraro & Pressey, 2015). We thus analyzed specific zoning units, areas, and types of forest cover changes. We tested the following hypotheses: (i) Different zoning categories have different impacts on forest cover loss and gain; (ii) protected areas cause leakage to neighboring areas; (iii) protected areas with an operational management plan, with stricter regulations, or that are long-established are more effective at reducing deforestation compared to others; and (iv) protected areas are more effective at deterring forest conversion for agriculture or timber extraction than forest loss due to forest fires and natural hazards.

2. Land-use zoning of forestlands in Bhutan

Managed according to customary laws in the past, forestlands were nationalized in 1969 under the Bhutan Forest Act. Although the first forest management plans were already implemented during the 1960s to limit timber extraction, this Act is the first national policy seeking forest protection, notably through patrolling by forest officers (Penjore & Rapten, 2004, pp. 21–27). It was replaced in 1995 by the Forest and Nature Conservation Act, which defined all forestlands as Government Reserved Forests, except for community forests and private forests that represented around 1% of forestlands in 2005 (FAO, 2014; Jadin et al., 2015; RGoB, 1995). A forest management plan is mandatory for land declared as Government Reserved Forests and no clearing for agriculture, setting fires, or removing forest produce is allowed, except for collecting products for domestic purpose with the proper permit (Penjore & Rapten, 2004, pp. 21–27; Dhital, 2009). Implementation of this legal framework was supported by the Forest and Nature Conservation Rules of 2000, 2003 and 2006 (DoFPS, 2011). These Rules specify land-use regulations, management, and related penalties for each type of forestland zoning units (RGoB, 1995; RGoB, 2006).

The first protected area (PA) of Bhutan, the Manas Game Sanctuary

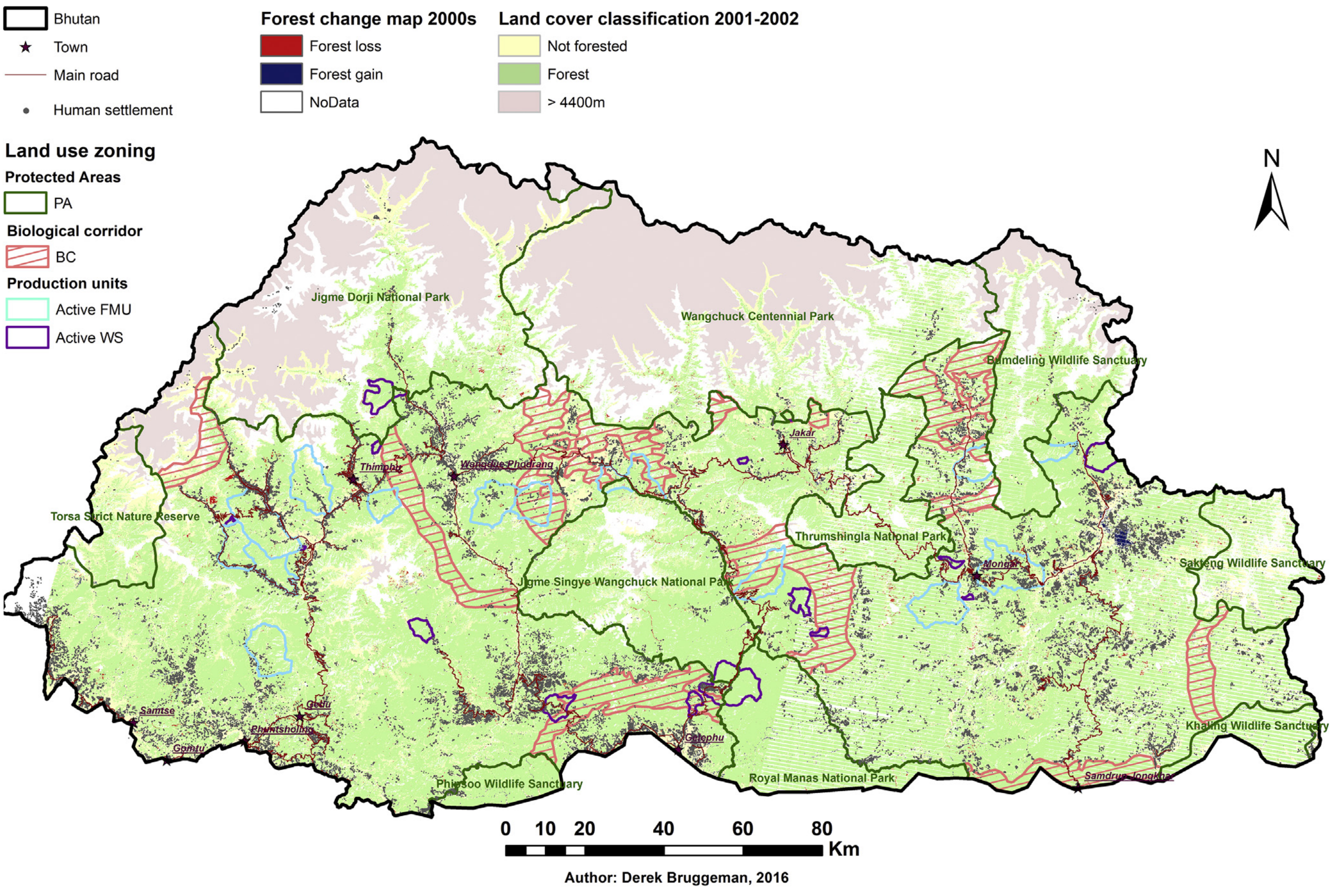

Fig. 1. Location and type of land zoning units in Bhutan. Note: The 2001–2002 land-cover classification and the 2000s forest change map (2001/2–2011) were taken from Bruggeman et al. (2016). They ignored areas at > 4400 m as this threshold corresponds to the upper timberline.

(latter upgraded as the Royal Manas National Park), was established in 1966 (Penjore & Rapten, 2004, pp. 21–27). It was followed by the Jigme Dorji National Park and the Khaling Wildlife Sanctuary (Table 1). Except for the newly created Wangchuck Centennial Park, other protected areas were all designated during the 1990s consecutive to the 1993 revision of the protected area system, to represent the different ecosystems of Bhutan (Tshering, 2003). To optimize biodiversity conservation and ecosystems representativeness (Tshering, 2003), protected areas are located across elevations and away from population centers (Fig. 1). The implementation of management plans was not considered operational for some protected areas during our study period (Table 1, Lham, Wangchuk, Stolton, & Dudley, 2018). These nature conservation units can also be differentiated according to stringency of rules and permissions regarding activities inside their boundaries, as defined by IUCN categories (Table 1) (Andam et al., 2013).

Biological corridors (BCs) connect one or more PAs for wildlife movement (RGoB, 2006; Wangchuk, 2007). Their spatial arrangement was defined to include areas with high forest and ground cover, limited human footprint, moderately rugged topography, and indications of movement of key wildlife species (GEF and UNDP, 2001). They join protected areas following the most direct path that avoids important human settlements and maintains high forest cover (Fig. 1). Officially recognized for protection since 1999, the system of biological corridors is still lacking effective land-use regulations (Brodie et al., 2016; GEF and UNDP, 2001). Yet, government incentives were introduced to limit tree cutting for domestic consumption (Wangchuk, 2007).

Forest management units (FMUs) are designated in operable areas for production. This comprised all Government Reserved Forests, except for forests in protected areas, in critical watersheds, or in a depleted or fragile state, such as forests on steep slopes or at high elevations (FAO, 1999; Winkler, 1999). These units are dedicated to timber production and must be managed following silvicultural practices adapted to forest type, as reflected in forest management plans (FAO, 1999; RGoB, 2006). These plans, running on a 10-year period, must ensure sustainable timber production notably through annual allowable cut limits and regeneration of harvested areas (FAO, 1999). However, the lack of financial and human resources was limiting the proper implementation of forest management plans at the turn of the 21th century (FAO, 1999). Production units are complemented by working schemes (WS), which are smaller units issued for a short period of time to respond to urgent or specific needs for timber to rebuild or create infrastructure. Little information exists about their management and spatial distribution.

Following neighboring Indian Himalayan States and Nepal, Bhutan initiated a community forest program in 1995 (Rahut, Ali, & Behera, 2015; Rasul, Thapa, & Karki, 2011; Somanathan, Prabhakar, & Mehta, 2009). After a slow start, half thousand community forests were approved in 2013, covering 567 km^2 in 2015 (Belsky, 2015; FAO, 2014). Of small sizes, they were mainly allocated in previously harvested forestlands, with therefore a high potential for forest regrowth.

3. Methodology

Evaluating impacts of a conservation policy (the treatment) consists of estimating the average treatment effect on the treated (ATT) for one or more impact variables (the outcome). This ATT represents the difference between the outcome for the treated and the outcome that would have been observed in treated areas had they not been subjected to the treatment – i.e., the counterfactual, which cannot be observed (Blackman, 2013). Valid inference on the ATT thus requires the credible estimation of the counterfactual. This can be achieved using statistical matching, which consists of pairing treated observations with similar control observations with respect to characteristics affecting treatment selection and the outcome – i.e., the confounding or control variables. Matching is now widely used to evaluate the impacts of multiple conservation policies for which explicit spatial distributions of these policies is available (Blackman, 2013; Meyfroidt, 2016).

3.1. Outcome variables

We defined two outcome variables: loss and gain in forest cover. We used a recent remote sensing study detecting forest cover changes based on Landsat imagery (Bruggeman et al., 2016). The study combined supervised classifications and image differencing on spectral indices for two time periods: 1990–2001/2 and 2001/2–2011. Due to clouds and snow, a small part of Bhutanese forests was ignored (Fig. 1). All geomatics treatments were performed in ArcGIS (ESRI, 2010).

We adopted a sampling strategy to match treated with control observations. We randomly selected 70,000 Landsat pixels, at 30-meter resolution (the unit of analysis), considering the low frequency of land-cover changes observed in the study area. We used the native spatial resolution of the Landsat data for the sampling. Given the small size of most forest cover change patches in Bhutan (Bruggeman et al., 2016), this scale was the most adequate to capture land use processes leading to forest cover changes (Avelino, Baylis, & Honey-Rosés, 2016). Sampling was constrained by a 100 m minimum distance between each observation to achieve a balance between minimizing spatial autocorrelation, as measured on variograms, and obtaining a sufficient number of forest cover change pixels. Among these pixels, 51,791 were forested in 2001/2 and constituted the set of observations for forest loss analyses. We used remaining points for forest gain analyses. 375 and 486 pixels experienced forest loss and forest gain respectively during the 2000s.

3.2. Treatment and control variables

Treatment variables refer to the different categories or subcategories of the land-use zoning of forestlands. For the PA category, we excluded observations located in the Wangchuck Centennial Park given its late establishment (Table 1). Because several production units in operation during the 2000s overlapped with BCs (Fig. 1), we made a distinction between all areas included in a BC and areas classified exclusively as BC. We also included FMUs as a treatment variable given their designation for sustainable management (RGoB, 2006) and because production units have proven to be effective at deterring forest loss in other countries, sometimes in similar or higher proportions than protection units (Bruggeman et al., 2015; Gaveau et al., 2013). We considered areas as treated if a FMU was operational for 3 years or more between 2001/2 and 2011 (mean = 6.3y) (SFC, 2012), which allowed including a sufficient number of treated observations. Active FMUs were mainly located in the northern half of the country (Fig. 1), where coniferous forests dominate. For WSs, we only had partial information regarding periods of activity. We selected operated and operational WSs as treatment variable (SFC, 2012). We could not test the effectiveness of community forests due to a lack of spatial data.

Each treated observation must be matched with an untreated (or control) observation. This control observation is selected by a matching algorithm in areas not subjected to a specific zoning. For all matching analyses, we removed from our random sampling the observations located inside the Wangchuck Centennial Park and inside FMUs in operation for less than 3 years, as these zoning units could have had impacts on forest cover trajectories.

Control variables – or covariates – included factors that were potentially influencing both forest cover changes and the location of zoning units: distance to main road, distance to town, distance to settlement, ecoregion, and aspect (Table 2). The first two variables reflect market accessibility while the third distance variable represents accessibility for local agents of deforestation. Given the ruggedness of the Bhutanese landscape, we modified these distance variables to integrate topography using Tobler's hiking function, which estimates travel time, accounting for the slopes of the raster cells crossed (Tobler, 1993). We hypothesized that ecoregion and aspect can confound treatment effect by modifying environmental conditions of a pixel and therefore its attractiveness for land conversion or its propensity to experience forest regrowth. Roads, towns and settlements locations in Bhutan are very stable, and mostly pre-date the establishment of protected areas, so that these variables are exogenous to the location of these protected areas.

For the forest loss outcome analyses (Table 2), we included travel distance to the forest edge as a control variable to reflect differentiation in ease of access to forest. We also added travel distance to forest lost during the previous period (1990 till 2001/2) as a covariable to account for potential omitted factors that could trigger forest loss and not be randomly distributed across treatment and control observations. This variable can account for time invariant unobservable characteristics, particularly if these characteristics had a decreasing effect with increasing distance from forest loss pixels. For example, a forest pixel could have a higher likelihood of being converted to agriculture if it has a high agricultural suitability. Neighboring pixels that are already cleared probably share some of these favorable conditions for agriculture. In contrast, this covariable will fail to account for time-varying unobservable variables that could affect forest cover dynamics, such as local introductions of new agricultural technology (soil amendments, crop variety, cow breeds …).

For the forest gain analyses (Table 2), we used the same covariates to find matched pairs, with two exceptions. Firstly, we replaced the travel distance to forest lost in the 1990s by the travel distance to forest gain during the same period. Secondly, we substituted the travel distance to forest edge by the proportion of forest cover in a 200 m radius around each observation. This variable represents the probability of a pixel to be recolonized by forest via seed rain. We chose a 200 m threshold as previous studies reported a dramatic decrease of seed rain and seed dispersal by animals at distances to forest edge between 80 m and 300 m (Carson & Schnitzer, 2008; Chazdon, 2014; Cramer & Hobbs, 2007, p. 449).

When necessary, we transformed continuous covariates to obtain a normal distribution. We did not include elevation and climatic variables as control variables because their possible influence on forest cover changes was already represented by other covariates (Table 2). We checked pre- and post-matching balance of these variables (called supplementary variables) to avoid large discrepancies between treated and control groups (Table 2).

3.3. Matching

An exact matching (identical values of all covariates for each matched pair) is not achievable in finite samples, particularly with many covariates and some of them being continuous (Sekhon, 2011). Here we requested an exact match for the ecoregion variable while we balanced for the other covariates using genetic matching. This algorithm achieved better balance across treated and control groups than classic propensity score or multivariate matching based on Mahalanobis distance (Canavire-Bacarreza & Hanauer, 2013; Diamond & Sekhon, 2013). We performed 1:1 nearest neighbor matching with replacement. All statistical analyses were performed in R (R Core Team, 2017).

Differences between treatment and control groups for specific covariates can persist post-matching. To test the quality of the matched pairs of observations for each covariate we used the standardized difference in means and the Kolmogorov-Smirnov (KS) tests. There is no generally-agreed threshold regarding the first statistic, with tolerance ranging from 0.1 to 0.4 (Brandt, Nolte, Steinberg, & Agrawal, 2014; Heilmayr & Lambin, 2016; Miranda, Corral, Blackman, Asner, & Lima, 2016). We decided to use Imbens and Wooldridge (2009) rule of

Table 2
Characteristics of control variables.

Variables	Abbreviation	Description	Unit	Confounding potential	Date of the data	Source of data
Common covariables						
Distance to main road	droads	travel distance by foot to reach/from[a] national or paved roads	hour	accessibility to markets	2006	NSB and the World Bank (2010)
Distance to town	dtown	travel distance by foot to reach/from[a] nearest town of at least 4000 inhabitants (Thimphu, Gedu, Samtse, Phuntsholing, Gelephu, Samdrup Jongkhar, Mongar, Jakar, Gomtu, Wangdue Phodrang)	hour	accessibility to markets	2005	OCC (2006)
Distance to settlement	dsettl	travel distance by foot to reach/from[a] nearest settlement	hour	accessibility for local agents of deforestation	2000–2012	Bhutan GeoSpatial Portal (2012) + own additions
Ecoregion	ecoregion	Global Ecological Zones present in Bhutan (from south to north): Tropical rainforest, tropical mountain system, Subtropical mountain system	class	local or regional environmental conditions	2010	FAO (2012)
Aspect	aspect	1 = north-facing slopes; 2 = west or east-facing slopes; 3 = south-facing slopes and flat areas	class	local or regional environmental conditions	2000	Jarvis, Reuter, Nelson, and Guevara (2008)
Covariables for forest loss						
Distance to forest loss 90s	dFloss	travel distance by foot to reach/from[a] nearest pixel which underwent forest loss between 1990 and 2001/2	hour	unobserved time invariant characteristics	2001/2	Bruggeman et al. (2016)
Distance to forest edge	dFedge	travel distance by foot to reach/from[a] the nearest group of unforested pixels (> 0.63 ha)	hour	accessibility for local agents of deforestation	2001/2	Bruggeman et al. (2016)
Covariables for forest gain						
Distance to forest gain 90s	dFgain	travel distance by foot to reach/from[a] nearest pixel which underwent forest loss between 1990 and 2001/2	hour	unobserved time invariant characteristics	2001/2	Bruggeman et al. (2016)
Proportion of forest in 200 m radius	pForest	% of forested pixels in 2001/2 in a radius of 200 m	%	local or regional environmental conditions	2001/2	Bruggeman et al. (2016)
Supplementary variables						
Elevation			meter	local or regional environmental conditions	2000	Jarvis et al. (2008)
Mean annual temperature			°C*10		1950–2000	Hijmans, Cameron, Parra, Jones, and Jarvis (2005)
Annual rainfall			mm		1950–2000	Hijmans et al. (2005)

[a] Travel distances by foot were calculated as the average of the time distance from the element potentially favoring forest loss (main road, settlement …) to the observation and the time of the reverse path using Tobler away and Tobler towards hiking's functions (Tobler, 1993).

thumbs of 0.25. We used significance at the 0.05 level for KS tests.

To overcome possible imbalance according to one or both of the metrics used, we first used post-matching bias adjustment, which removed some of the remaining bias through linear regression on the covariates (Imbens, 2015). If necessary, we improved balance by removing matches outside the tolerance level defined by a caliper – i.e., distance between matched observations for each covariate, expressed in standard deviations (Andam et al., 2013) – to obtain covariate distributions that were statistically undistinguishable among the treated and the control groups. As this operation drops matched pairs that are too dissimilar, the difference between the outcomes of treated and control observations may no longer represent the overall ATT but rather the average treatment effect on part of the treated (Andam et al., 2013; Arriagada, Ferraro, Sills, Pattanayak, & Cordero-Sancho, 2012). In cases where < 75% of the pairs were kept, we plotted the distribution of distance covariates and elevation for the full treated sample (matching without caliper) and for the reduced treated sample (matching with caliper) to visualize the impact of sample reduction (Appendix B). We calculated heteroscedasticity robust standard errors (Abadie & Imbens, 2006) to evaluate precision of our estimates (Ferraro et al., 2013).

Matched pairs are thus comparable according to observed characteristics potentially influencing the probability of being assigned to treatment and affected by forest cover change. However, variables having an influence on these two probabilities could have been omitted or be unobservable. We tested the sensitivity of our matching results to these potential unobserved covariates by computing Rosenbaum bounds (Rosenbaum & Rubin, 1983). This sensitivity test simulates non-equal odds of treatment for the treated observations compared to the control group due to an unobserved factor. By increasing the odds – the gamma value (Γ) –, we can find at what influence of unobserved bias our inference would be invalidated (Canavire-Bacarreza & Hanauer, 2013). Finally, when we compared the effectiveness of two different types of zoning or of contexts (see hypotheses below), we tested the significance of the difference of the effects using the Chow test (Chow, 1960; Zeileis, Leisch, Hornik, & Kleiber, 2001).

3.4. Effectiveness assessment

We tested hypotheses regarding the ability of land-use zoning to protect (for the forest loss outcome) or enhance (for the forest gain outcome) forest cover in Bhutan (Table 3).

Hypothesis 1. We expected different impacts on outcome variables between each zoning category given their different purposes and land-use regulations. Specifically, we anticipated a reduction in forest loss inside PAs and, to a lesser extent, in FMUs. PAs are likely to have increased forest gain following the 1995 ban on shifting cultivation, which was particularly enforced in PAs (Namgyel, Siebert, & Wang, 2008). In FMUs, restrictions on agricultural practices could be compensated by natural or planted forest regrowth in previous timber exploitation plots. Considering that BCs were not yet effectively managed, we expected similar forest loss and gain than in the control group. By responding to urgent need for timber without long-term management, we expected WSs to display higher forest loss and similar forest gain compared to the counterfactual.

Hypothesis 2. We expected PA protection to cause leakage to neighboring areas. This was tested by matching forested areas within a 5 km buffer around PAs with remaining forestlands outside zoning. We also analyzed the impact of PA on forest loss by excluding this buffer from potential control observations.

The average effectiveness of the PA network might hide heterogeneities in achieving their forest conservation objectives. This can be due to variations in stringency, enforcement and duration of treatment (Table 1); in forest type subjected to treatment; and in causes of forest cover changes. Coniferous and broadleaf forests exhibited asymmetric rates of changes in the 2000s, with higher gross forest gain in the broadleaf forest ecosystem due to different historical shifting cultivation systems (Bruggeman et al., 2016). An identical policy instrument can perform well at deterring forest fires for instance, but poorly at avoiding deforestation for infrastructure extension. Similarly, deforestation for agriculture could be effectively curtailed in a long-established PA while ignored in a more recent PA. We performed several matching analyses based on subsamples of our observations to test the following hypotheses (Table 3).

Hypothesis 3. We expected higher avoided forest loss for PAs with an operational management plan (hereafter called 'operational PAs'), PAs with stricter regulations (IUCN category I or II, called 'strict PAs'), and long-established PAs (created before 1975, called 'old PAs') (Table 1). We also tested the performance of operational PAs and PAs created during the 90s ('new PAs') for forest gain, as it could be particularly high just after protection.

Hypothesis 4. We expected PAs to be more effective at deterring forest conversion for agriculture or timber extraction than forest loss due to forest fires and natural hazards. While there is plenty of forestlands outside zoning units susceptible to fulfill land demand associated with agriculture and logging activities, and shifting cultivation is being gradually replaced by intensive agriculture (Bruggeman et al., 2016; Namgyel et al., 2008), forest loss caused by fires, glacier lake outburst or flash floods is more difficult to avoid inside PAs. To test this hypothesis, we ran several matching analyses restricted to operational PAs (Table 3). First, we differentiated pixels located in broadleaf and in coniferous forests as they experienced different forest change dynamics. In particular, coniferous forests are more prone to fires. Secondly, we focused on PAs' ability to limit loss in forested areas that are suitable for agriculture. We restricted these areas to forests close to human settlements (less than 30 min by foot) or main roads (less than 20 min), and below elevations and slopes above which agriculture is unlikely. According to the Bhutan Land Cover Assessment 2010 (NSSC and PPD, 2011), less than 1% of north-facing agricultural lands were located at elevations higher than 3000 m or slopes steeper than 35°. For other slope aspects, elevation increased to 3250 m and slopes to 40° (west or east-facing) or 42° (south-facing). We selected these slope and elevation values as thresholds. Thirdly, we isolated forest fires and natural hazards of significant size by selecting only forest loss patches: (i) larger than 18 pixels, a natural break point in the distribution of forest patch sizes, and (ii) located at path distances of more than 30 min from settlements and more than 20 min from main roads, to exclude large forest clearings for agriculture or infrastructure extension.

4. Results

4.1. Pre-matching forest cover changes in land zoning units

During the study period, rates of gross forest loss in PAs, BCs, FMUs and WSs were lower but comparable to rates measured in forestlands not under a specific zoning (Fig. 2). These differences in rates represent naïve estimates of the impact of zoning as they ignore the selection bias. Protection units (PAs and BCs) exhibited higher rates of forest loss than units allocated to forest production (FMUs and WSs). Gains in forest areas in non-restricted forestlands were almost three times as high as rates detected in each type of zoning unit during the 2000s (Fig. 2). In each category, net forest change in the 2000s was higher than in the 1990s, except for FMUs. Non-gazetted forests shifted from net forest loss to net forest gain.

Within PAs, rates of forest loss were more variable in the core zone of PAs than close to their boundaries (Fig. 3). The relation between forest loss and distance to unprotected areas was not significant however ($p = 0.128$). We observed a marked increase in forest loss when crossing the boundary to enter areas without protection (Fig. 3). In these forestlands outside zoning units, rates of forest loss were

Table 3
Hypotheses and related effectiveness assessments performed.

Hypothesis			Universe		Outcome	
No.	Rationale	Test	Treatment	Control	Forest loss	Forest gain
1	Effectiveness varies across the different categories of land zoning, and for different directions of forest cover change. Avoided forest loss is high in PAs, moderate in FMUs, and absent in BCs, and more losses are observed in WSs. PAs also cause more forest gain.	PA	PA	Land outside zoning	0 = unchanged forest 1 = forest loss	0 = unchanged non forest 1 = forest gain
		FMU	Active FMU	Land outside zoning		
		All BC	Biological corridor			
		Only BC	Only biological corridor			
		WS	Active WS			
2	Effectiveness of PAs drives leakage in surrounding forestlands.	Buffer PA	Land at < 5 km from PA	Land outside zoning and at > 5 km from PA	0 = unchanged forest 1 = forest loss	/
		PA (buffer excluded)	PA			
3	Effectiveness of PAs depends on the characteristics of the units, with higher effectiveness in areas (i) with stricter regulations, (ii) long-established, especially for forest gain, and (iii) with operational management plans.	Old PA	PA created < 1975	Land outside zoning	0 = unchanged forest 1 = forest loss	0 = unchanged non forest 1 = forest gain
		New PA	PA created > 1992			
		Old PA (vs new PA)	PA created < 1975	PA created > 1992		
		Op. PA	Operational PA	Land outside zoning		
		Strict PA	PA of category I or II	Land outside zoning		
4	Effectiveness of the zoned units vary for different mechanisms of forest cover change. PAs are more effective at deterring forest loss originating from conversion to agriculture or timber extraction than from forest fires or natural hazards.	Op. PA (broadleaf)	Operational PA	Land outside zoning	0 = unchanged broadleaf forest 1 = forest loss in broadleaf forest	/
		Op. PA (conifer)			0 = unchanged coniferous forest 1 = forest loss in coniferous forest	
		Op. PA (pot. culti.)			0 = unchanged potentially cultivable forest 1 = forest loss in potentially cultivable forest	
		Op. PA (fire)			0 = unchanged forest away from human influence 1 = large forest loss away from human influence	

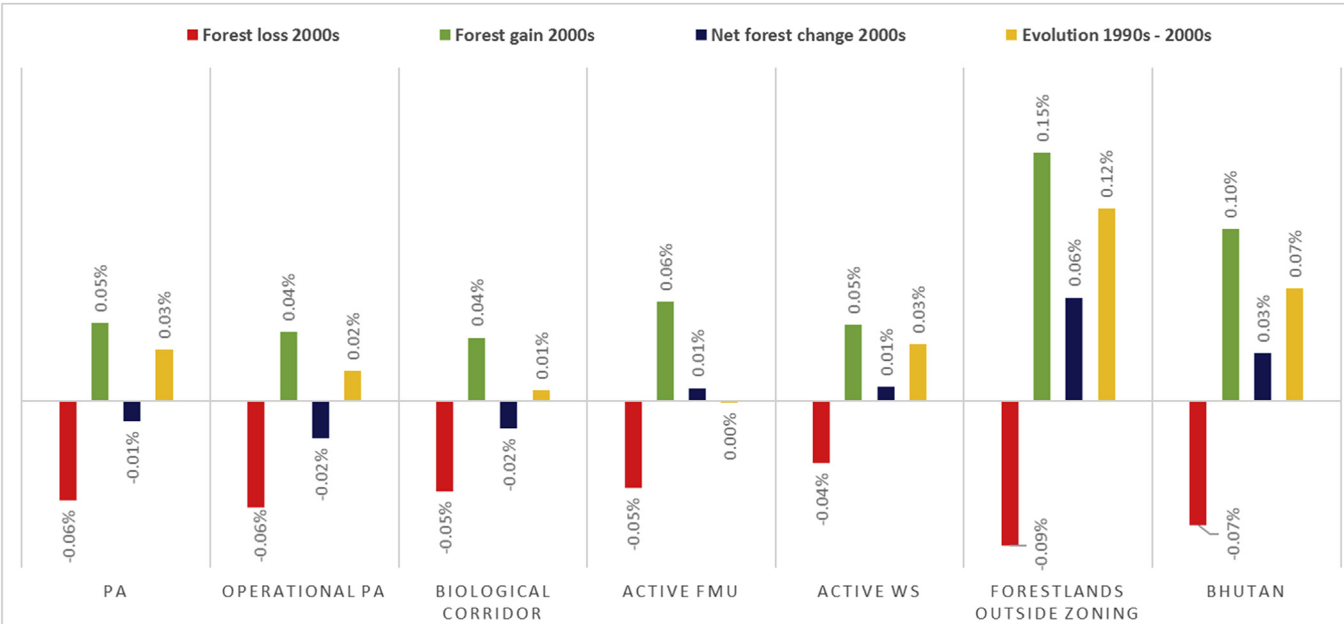

Fig. 2. Annual rates of forest conversion by type of zoning units during the 2000s (between 2001/2 and 2011). Note: The 'Evolution 1990s - 2000s' histogram bar refers to the net annual forest change rate during the 2000s minus the net annual forest change rate during the previous decade (from 1990 till 2001/2).

decreasing with increasing distance to PAs ($p < 0.01$).

4.2. Biases in the location of zoning units

Elevation and distances to main roads and town of PAs were only moderately higher than that of forest areas not subjected to zoning (Fig. 4A). Forest areas inside PAs were however clearly farther from forest edges and human settlements. Remoteness and high elevation were exacerbated for non-forested areas under protection (Fig. 4B). The tropical mountain forest domain, where human population concentrates, was underrepresented in the PA network. Siting of BCs was similar to that of PAs, but with smaller mean values of the distance covariates. Forestlands of BCs are closer to towns and main roads than forestlands located outside zoning (Fig. 4). FMUs were located closer to human landscape features generally associated with forest loss (settlements, town and main roads) compared to forested areas outside zoning units (Fig. 4A). Absence of FMUs in the southern part of the country – corresponding to the tropical rain forest ecoregion – explained the higher elevation and lower rainfall of forestlands in FMUs. WSs were also situated at short distances from a main road.

4.3. Differences in forest cover changes after matching

Hypothesis 1. Results presented here are for the matchings with covariate balance achieved for both our metrics – i.e., standardized difference in means and Kolmogorov-Smirnov tests –, thus with calipers for several analyses. The differences in proportions of forest loss between treated and control observations for each type of treatment (Fig. 5) reveal that PAs significantly reduced forest loss during the 2000s. The rate of forest loss dropped from 0.97% outside of protected areas to 0.36% within these, thus decreasing by 0.59 percentage points. Although it only applies to 46% of treated observations, this post-matching relative effect on forest loss was higher than the relative effect from the naïve estimate (Appendix A).

Despite their timber extraction objectives, active FMUs were associated with a much lower forest loss, which was halved by this treatment ($p < 0.05$). Biological corridors ('All BCs') reduced forest loss compared to similar areas not located in BCs. When excluding FMUs and WSs from BCs ('Only BC'), the ATT of BC was not statistically different from zero (Fig. 5). Hence, the effects of BCs hinged on the effectiveness of FMUs in reducing forest loss. We could not detect an

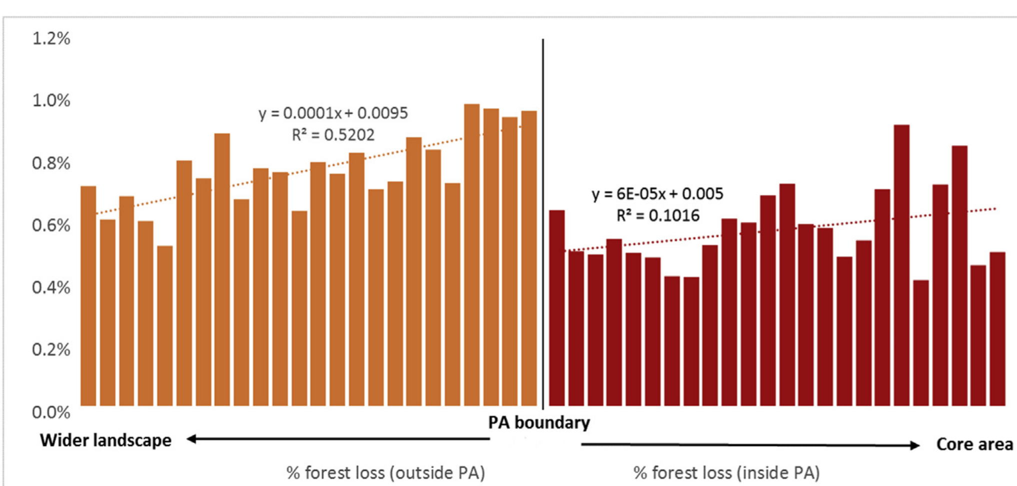

Fig. 3. Rates of forest loss between 2001/2 and 2011 inside and outside PAs according to the path distance to PA boundaries. Notes: In PAs, each histogram bar represents 1/25 of the area located inside the PA network (excluding Wangchuck Centennial Park given its recent creation). The same time distance thresholds were used for area located outside PA boundaries.

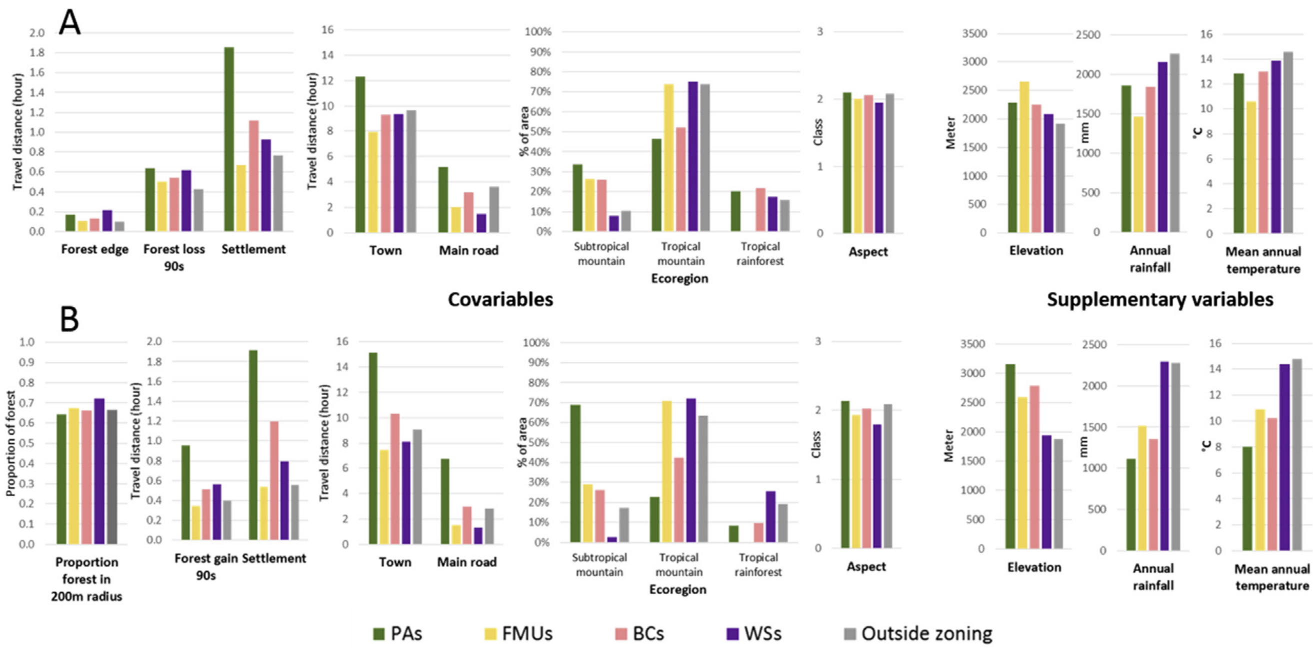

Fig. 4. Mean location of (A) forestlands and (B) non-forested lands in zoning units and in areas outside zoning. Note: See Table 2 for description of covariables and supplementary variables.

influence of WSs on forest loss but only few treatment points were used (Appendix A).

Contradicting Hypothesis 1, we found significantly lower post-matching proportions of forest gain in PAs compared to control areas ($p < 0.05$) (Fig. 6). In agreement with our expectations, FMUs, BCs and WSs had comparable proportions of forest gain than paired untreated observations (Fig. 6).

Hypothesis 2. After matching, we observed that being located around PAs increased the probability of facing forest loss by 0.34 percentage points ($p < 0.01$) (Fig. 5). This represents circa twice as much forest loss than in forestlands distant from PAs (Appendix A), and suggests a potential leakage of forest loss around PAs. The PA effectiveness analysis excluding buffer areas from the potential control points did not affect the results on overall effectiveness of PAs, but decreased their relative effect on forest loss from 63% to 56%. This difference, however, was not statistically significant under the Chow test for matching with calipers (p-value = 0.290, Appendix C).

Hypothesis 3. In agreement with Hypothesis 3, long-established PAs reduced forest loss by 1.26 percentage points ($p < 0.01$) (Fig. 5), corresponding to a relative effect on forest loss of 85% (Appendix A). Newly created PAs also significantly decreased forest loss, but in smaller proportions compared to old PAs or all PAs. Pairing forested areas inside old PAs with similar forestlands inside new PAs highlighted the superior effectiveness of long-established compared to recent PAs (Fig. 5). This difference was statistically significant under the Chow test (p-value = 0.0405, Appendix C). Note that the strict caliper retained

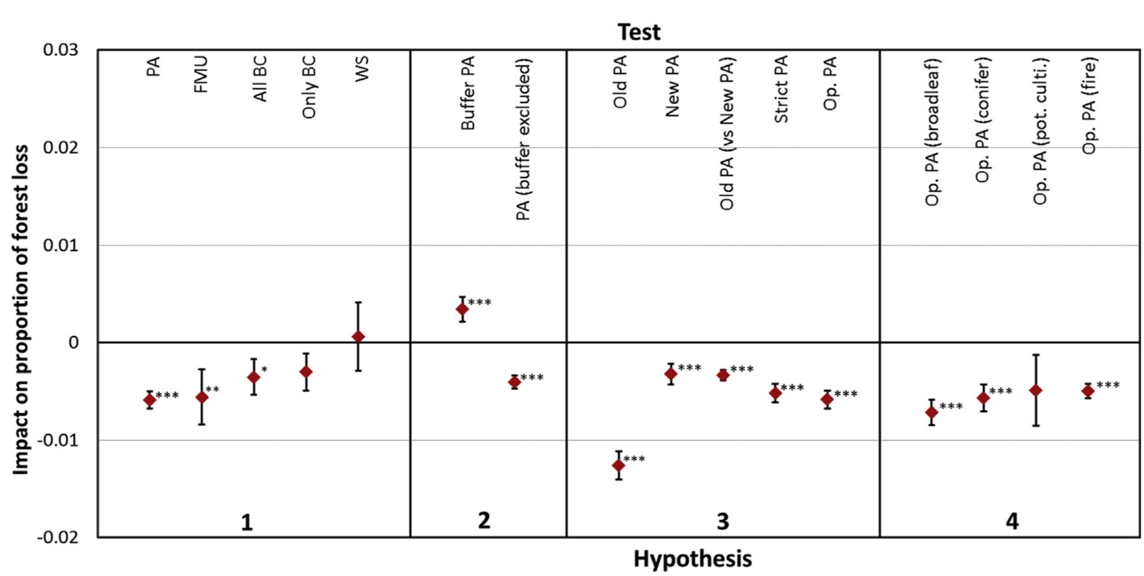

Fig. 5. Treatment effects on forest loss after genetic matching with bias adjustment. Notes: *Significance at the 0.1 level, ** significance at the 0.05 level and *** significance at the 0.01 level. Abadie and Imbens (2006) standard errors in bars.

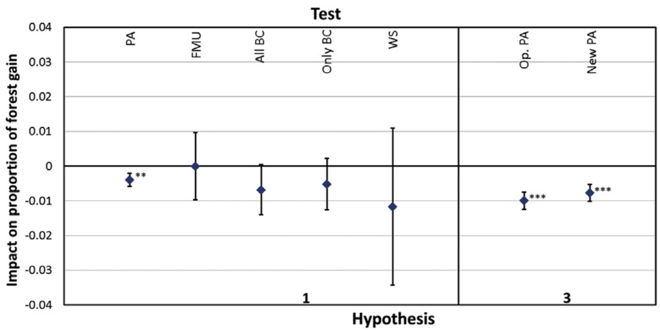

Fig. 6. Treatment effects on forest gain after genetic matching with bias adjustment. Notes: *Significance at the 0.1 level, ** significance at the 0.05 level and *** significance at the 0.01 level. Abadie and Imbens (2006) standard errors in bars.

only 13% of matched pairs (Appendix A). Strict PAs generated comparable avoided forest loss than all PAs (Fig. 5). Operational PAs – which represented 86% of forestlands under protection - showed almost identical results than those obtained for all PAs (Fig. 5), with a significant avoided forest loss.

For the forest gain outcome, operational and recent PAs were not associated with forest regeneration during the 2000s, as the treatment effect was a significant decrease in forest gain (Fig. 6).

Hypothesis 4. We measured somewhat comparable effectiveness when restricting our analysis to coniferous or broadleaf forests inside operational PAs, with a very highly significant average treatment effect on the treated (Fig. 5). The relative effect on forest loss was estimated at 79% in broadleaf forests and 59% in conifer forests. The Chow test indicates that these effects are significantly different (p-value = 0.00192, Appendix C), with effectiveness of the protection being higher on broadleaf forests. For potentially cultivable forestlands, forest loss between treated and controls was not significantly different (45%) (Fig. 5). After restricting our outcome variable, we found a very highly significant effect of operational PAs on forest loss in large patches far from human influence, with a 71% reduction in forest loss.

Sensitivity analysis

Our results were quite sensitive to potential unobserved covariates. Statistically significant impacts on forest loss or forest gain were nullified (at the 0.05 level) for values of Γ – i.e., the ratio between probability of treatment for the treated and control group – inferior to 2 for each matching analysis, except for the estimate of long-established PAs (Appendix A). Applying a caliper greatly affected the sensitivity of statistical results by increasing resistance to unobserved heterogeneity among matched pairs (Appendix A).

5. Discussion

5.1. Effectiveness of protected areas and other zoning units

Tshering (2003) assessed the management effectiveness of four operational PAs in Bhutan in the early 2000s based on interviews with park managers, staff and stakeholders, following the Rapid Assessment and Prioritization of Protected Area Management (RAPPAM) Methodology. Interviewees stated that PAs were mainly threatened by poaching, grazing and road construction, with shifting cultivation, timber felling and forest fires considered as minor to non-existent threats (Tshering, 2003). A development program was initiated to support PAs residents and limit their environmental impacts through financial incentives (Wangchuk, 2007). Except for a significant reduction in firewood collection, little evidence exists on the environmental benefits of this program. Shifting cultivation and livestock rearing remained the main activities of PAs residents (Rinzin, Vermeulen, Wassen, & Glasbergen, 2009; Wangchuk, 2007). PAs stakeholders acknowledged that law enforcement and systematic monitoring of impacts of legal and illegal uses inside PAs were inadequate or absent due to insufficient staff and equipment (Tshering, 2003). A recent study, based on interviews of PAs staff, confirmed that these deficiencies hamper proper implementation but concludes that, under the current conditions of low human pressure, the PAs network effectively attains the management objectives (Lham et al., 2018).

We observed a significant and high percentage of avoided forest loss within compared to outside PAs (in agreement with Hypothesis 1), which agrees with findings for other protected are PAs as networks in tropical regions (Carranza, Balmford, Kapos, & Manica, 2014). The number of treated observations located far (> 3 h) from human settlements was drastically reduced from our impact evaluation of PAs (Appendix B). We therefore concluded that PAs were effective at deterring forest loss in accessible to moderately accessible locations. By contrast, zoning had limited impact on forest loss in core zones of PAs (Fig. 3).

Our results suggested that PAs effectiveness has triggered a leakage of forest loss in surroundings areas (in agreement with Hypothesis 2). This argues for greater control of these areas, notably through buffer zones around PAs and biological corridors (Rinzin et al., 2009; Tshering, 2003).

Avoided forest loss estimated for strict and operational PAs were comparable to that of all PAs (contrary to Hypothesis 3). However, old PAs appeared more effective at curtailing forest loss than recent ones. This result was however hindered by the limited number of well-balanced matched pairs (12% of all points inside old PAs).

PAs greatly reduced large forest loss patches away from human influence (contrary to Hypothesis 4), although this unconfounded estimate was not valid for large forest losses located at long distance from human footprints (Appendix B). PAs were apparently less effective at reducing forest loss inside fire-sensitive conifer compared to broadleaf forests. These protected units did not significantly reduce forest loss in potentially cultivable forestlands, a probable consequence of the limited number of forest loss observations (Fig. 5; Appendix A). The lower than expected PA impact on forest conversion to agriculture could be associated with the weak support by local communities towards PA establishment, which was only gradually improving during the study period thanks to environmental education activities (Tshering, 2003). Shifting cultivation was also believed to be still practiced in the mid-2000s in at least one of the six operational PAs (Rinzin et al., 2009). This analysis suggests that PA effectiveness varies in part with the causes of forest loss, with a higher effectiveness in reducing anthropogenic deforestation in accessible areas compared to wildfires in remote, coniferous forests.

Biological corridors were officially designated just before our study period. As their management was not yet initiated in the 2000s (Wangchuk, 2007), we hypothesized and actually observed an absence of treatment effect on forest cover change (Figs. 5 and 6).

According to Dhital (2009), total area under forest management units might be reduced in the future due to land-use change taking place inside these units. Our results did not support this prediction as gross rates of forest cover changes were low inside these units (Fig. 2): they avoided 50% of forest loss inside their boundaries (Fig. 5).

Evidence on effects of the community forest program on forest resources protection, including tree cover evolution, remains scarce. Timber harvesting has been evaluated based on field inventories in specific community forests (Buffum, Gratzer, & Tenzin, 2009, 2008; Moktan, Norbu, & Choden, 2016) and appeared to be sustainable. Further studies are needed to analyze impacts of the CF program at the national scale.

5.2. Methodological considerations and scope of our impact evaluation

Estimates of treatment effect after covariates matching require that all the potentially confounding covariates are included. In our case, no longer active FMUs could have impacted forest gain during our study period, possibly leading to a downward bias in the effectiveness of the current zoning on forest gain. Potential remaining bias in our estimates was accounted for through the calculation of Rosenbaum bounds. Covariate values must be independent of the treatment. Ideally, covariate values must be measured before the treatment period or as close as possible to the start of the study period (Ferraro & Hanauer, 2015) – a condition that was largely met for this study. Thirdly, dates of outcome and treatment variables must coincide. For FMUs, treatment was only present during part of the study period in some cases, possibly leading to a downward bias in the assessment of their impacts. The context of Bhutan adds a challenge to effectiveness assessment of forest conservation policies as forests experienced limited land-cover changes during the study period (Bruggeman et al., 2016).

Except for PAs, our effectiveness assessment of land-use zoning categories is robust as an adequate balance across covariates was reached for the sample of treated observations. For PAs, an inadequate post-matching balance can be explained by a very strong selection bias, their extensive coverage, and their large size. Few areas far from human influence remained outside zoning units, making core zones of protection units challenging to pair. By design, protection units were established in sparsely populated areas. This low population density could also have been reinforced by forest protection.

The remaining covariate imbalance could confound the estimation of environmental impacts of a given intervention, and was addressed in two ways. First, we used post-match regression bias adjustment to control for bias resulting from imperfect matching. Secondly, we removed poorly matched pairs from our analysis using calipers until we obtained satisfactory balance. We observed large differences between impact estimates measured with and without caliper (Appendix A). In the non-caliper situation, the residual post-matching imbalance between covariates may have blurred the effect of land zoning. Further, this suggests that the effectiveness of land zoning was not homogeneous across all treated areas, with areas retained by the caliper having specific characteristics and thus reacting differently to the treatment. When a caliper is applied and drops a considerable share of treated units, impact estimates need to be interpreted in light of the new distribution of treated units that are actually tested. The estimates obtained after elimination of the imbalance (i.e., matching with caliper) only reflected the effects of land zoning on a subset of the area affected by the intervention, i.e., on areas with shorter distances to human settlements and main roads, and lower elevation compared to the full sample. Hence, we characterized the treated areas for which an unconfounded effect was estimated by comparing the covariates distribution of treated observations without and with caliper (Appendix B). For the analyses using calipers, our results may not hold for the parts of zoned units that are the most remote from human activity and at very high elevation. In these high elevation and remote areas, which are almost all under protection status, some large tree loss events likely resulting from wildfires may explain the higher likelihood of forest loss (Bruggeman et al., 2016). Yet, our estimates with calipers appropriately capture the effects of PAs on anthropogenic deforestation in the non-core parts of the PAs.

6. Conclusion

The Bhutanese network of protected areas, covering a substantial fraction of the country, effectively protected forest cover, with an effectiveness estimated at 63% of avoided forest loss. Yet, this represented a decrease of only 0.06% in the deforestation rate, from 0.097 $\%y^{-1}$ to 0.036 $\%y^{-1}$. Although protected areas in Bhutan contribute to forest protection in a statistically significant way, their contribution to avoided deforestation is limited given the low baseline levels of forest cover changes. Conversely, the low deforestation pressure in Bhutan makes it politically easy to expand the network of protected areas. We showed that the effectiveness of protected areas was higher for long-established protected areas and for the less fire-sensitive broadleaf forests. We also found evidence of leakage of forest loss from protected areas to their periphery. Our results also demonstrated that forest management units, which are dedicated to sustainable timber management, were effective at reducing half of the forest loss that would have taken place without these units. Forest conservation outcomes of logging units were already attested in other tropical regions. Biological corridors had no impact on forest cover change. Management of these corridors should ensure that their function of wildlife movement facilitation can be maintained in the long run.

During the 2000s, forestlands outside zoning units were characterized by limited gross forest loss, despite leakage in areas neighboring protected areas, and relatively high gross forest gain. These areas were responsible for the net forest gain at the national scale. This suggests that the land-use zoning implemented by Bhutanese authorities is embedded in a broader context of effective forest conservation policies at the national scale.

Acknowledgments

This study contributes to the Global Land Programme https://glp.earth.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.apgeog.2018.04.011.

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