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ORIGINAL ARTICLE

Evaluating whether protected areas reduce tropical deforestation in Sumatra

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ABSTRACT

Aim This study determines whether the establishment of tropical protected areas (PAs) has led to a reduction in deforestation within their boundaries or whether deforestation has been displaced to adjacent unprotected areas: a process termed neighbourhood leakage.

Location Sumatra, Indonesia.

Methods We processed and analysed 98 corresponding LANDSAT satellite images with a *c.* 800 m^2 resolution to map deforestation from 1990 to 2000 across 440,000 km^2 on the main island of Sumatra and the smaller island of Siberut. We compared deforestation rates across three categories of land: (1) within PAs; (2) in adjacent unprotected land lying with 10 km of PA boundaries; and (3) within the wider unprotected landscape. We used the statistical method of propensity score matching to predict the deforestation that would have been observed had there been no PAs and to control for the generally remote locations in which Sumatran PAs were established.

Results During the period 1990–2000 deforestation rates were found to be lower inside PAs than in adjacent unprotected areas or in the wider landscape. Deforestation rates were also found to be lower in adjacent unprotected areas than in the wider landscape.

Main conclusions Sumatran PAs have lower deforestation rates than unprotected areas. Furthermore, a reduction in deforestation rates inside Sumatran PAs has promoted protection, rather than deforestation, in adjacent unprotected land lying within 10 km of PA boundaries. Despite this positive evaluation, deforestation and logging have not halted within the boundaries of Sumatran PAs. Therefore the long-term viability of Sumatran forests remains open to question.

Keywords

Confounding variables, conservation, deforestation, Indonesia, leakage, propensity score matching, protected areas, Siberut, Sumatra.

INTRODUCTION

Most governments consider that establishing protected areas (PAs) and restricting human activities within PAs represents the best strategy for reducing tropical deforestation and conserving the intrinsic biodiversity of tropical forests (Leader-Williams *et al.*, 1990; Chape *et al.*, 2005). An estimated 23% of the Earth's humid tropical forest biome is now under protection to conserve biodiversity (UNEP/WCMC, 2007). Reducing tropical deforestation is also central to strategies for mitigating climate change (IPCC, 2007). Failure to do so could result in an additional release of 87–130 gigatonnes of carbon into the atmosphere by 2100, which would correspond to all the carbon released during the last decade through combustion of global fossil fuels (Gullison *et al.*, 2007). Therefore, the conservation of tropical forests through the establishment of PAs warrants a sense of urgency, both to reverse climate change and to conserve biodiversity.

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After decades of investment in efforts to protect tropical forests, many tropical terrestrial PAs have lost much of their natural forests through logging, conversion to agriculture and settlement (Curran *et al.*, 2004; DeFries *et al.*, 2005; Southworth *et al.*, 2006), raising questions over whether tropical PAs have been effectively implemented. It is often reported that PAs have not been adequately demarcated, are underfinanced and are staffed by underpaid guards who are consequently often prone to corrupt practices (Terborgh *et al.*, 2002), and that political leaders around the world favour more profitable extractive forest industries, such as logging and agricultural expansion, over conservation, particularly in biodiversity-rich accessible lowland forests (Laurance, 2001; Curran *et al.*, 2004; Soares-Filho *et al.*, 2006; Fitzherbert *et al.*, 2008). Furthermore, where tropical PAs are established, they are often thought to displace deforestation to adjacent unprotected areas, for example by attracting migrants and development projects in adjacent lands (Armsworth *et al.*, 2006; Wittemyer *et al.*, 2008), by relocating indigenous communities from PAs to adjacent areas (Brockington & Igoe, 2006) or by pre-emptive clearing of forest by landowners around newly created restricted-use areas (Oliveira *et al.*, 2007). This process, whereby deforestation increases along PA boundaries, in areas that would have otherwise remained undisturbed, we have termed 'neighbourhood leakage'.

Many now recognize the need to measure the effectiveness of conservation strategies to justify future investments in conservation (Mace *et al.*, 2008). Ferraro & Pattanayak (2006) have proposed using a clear indicator of human impact, such as fine-scale maps of tropical deforestation, coupled with an experimental design to predict the deforestation that would have been observed had PAs not been established. Assessments of the effectiveness of PAs have compared the intensity of human impact inside and outside PAs. Lower deforestation rates inside PAs than outside prompted the conclusion that PAs have been partially effective at conserving biological diversity (Bruner *et al.*, 2001; Naughton-Treves *et al.*, 2005). However, the simple inside–outside comparisons that have been used to date to assess the effectiveness of PAs (Bruner *et al.*, 2001; Sanchez-Azofeifa *et al.*, 2003; Naughton-Treves *et al.*, 2005; Nepstad *et al.*, 2006; Nagendra, 2008) may have considerably overestimated their protection effect, according to one recent study in Costa Rica (Andam *et al.*, 2008). Such comparisons ignore: (1) the role PAs might play in giving the impression that they are reducing deforestation when in fact they displace losses across space, for example through 'neighbourhood leakage'; and (2) the generally remote locations in which PAs are established, making PA lands less likely to be cleared, even in the absence of protection (Leader-Williams *et al.*, 1990; Pressey *et al.*, 1993; Joppa *et al.*, 2008).

The Indonesian island of Sumatra is a case in point. Sumatra has recently attracted global attention (IUCN, 2008) because it contains extensive biodiversity-rich lowland forests (Whitten *et al.*, 1987) and is estimated to hold one of Southeast Asia's largest stores of carbon (Page *et al.*, 2002; Uryu *et al.*, 2008), but is experiencing alarming rates of deforestation (Jepson *et al.*, 2001; FWI/GFW, 2002). Surprisingly, the effectiveness of Sumatran PAs has not been evaluated using appropriate quantitative indices, such as rates of deforestation. Reported deforestation rates calculated for Sumatra locally and for Sumatran PAs come with strong caveats (Scotland *et al.*, 1999). The current standard map of deforestation for Indonesia (FWI/GFW, 2002) lacks consistency at a fine spatial scale, for example a village, a small PA or subdistrict level (MoF, 2006), because the methods and definitions of 'forest' and 'deforestation' have varied during the time period considered. The global forest assessments of the UN Food and Agriculture Organization (FAO) present similar limitations (Olander *et al.*, 2008). Consequently, this study aimed to better evaluate whether PAs reduce deforestation in Sumatra by: (1) producing the first island-wide fine-scale map of deforestation, using a consistent methodology and clear definitions of what constitutes 'forest' and 'deforestation'; (2) testing for the possibility of 'neighbourhood leakage'; and (3) taking into account the generally remote locations in which Sumatran PAs are established. Analyses were performed using the statistical methods of propensity score matching and linear regression, both of which can remove the effects of confounding variables in robust ways (Rubin, 1973; Rosenbaum, 2005). Both methods provided much the same conclusions. Therefore, only the results of the propensity score matching are presented, given that this method makes very few parametric assumptions about the underlying structure of the data (Rubin, 1973; Rosenbaum & Rubin, 1985).

MATERIALS AND METHODS

Mapping tropical deforestation

We processed and analysed 98 corresponding LANDSAT TM and ETM+ satellite images with a *c.* 800 m^2 (28.5 × 28.5 m) resolution to map forest cover change from 1990 to 2000 across a total of 440,000 km^2 on the main island of Sumatra and the smaller island of Siberut. 'Forest' refers to old-growth natural evergreen forest (canopy cover > 50%), either undisturbed or partially degraded by selective logging. This definition corresponds closely to those forests defined by the FAO as closed broadleaved forests (FAO, 1993; Achard *et al.*, 2002). Second-growth forest was classified as non-forest because it was impossible to detect this land-cover class without extensive ground data, which were lacking. Therefore, forest regrowth from 1990 to 2000 was not quantified. Forest cover change only describes forest cover loss, i.e. deforestation. 'Deforestation' is the long-term removal of old-growth natural evergreen forest cover. Processing methods (which are similar to those used to map deforestation in Madagascar; Harper *et al.*, 2007) and map validation procedures can be found in Appendix S1 in Supporting Information. A map of 1990 forest cover and of 1990–2000 deforestation across Sumatra is presented in Fig. 1.

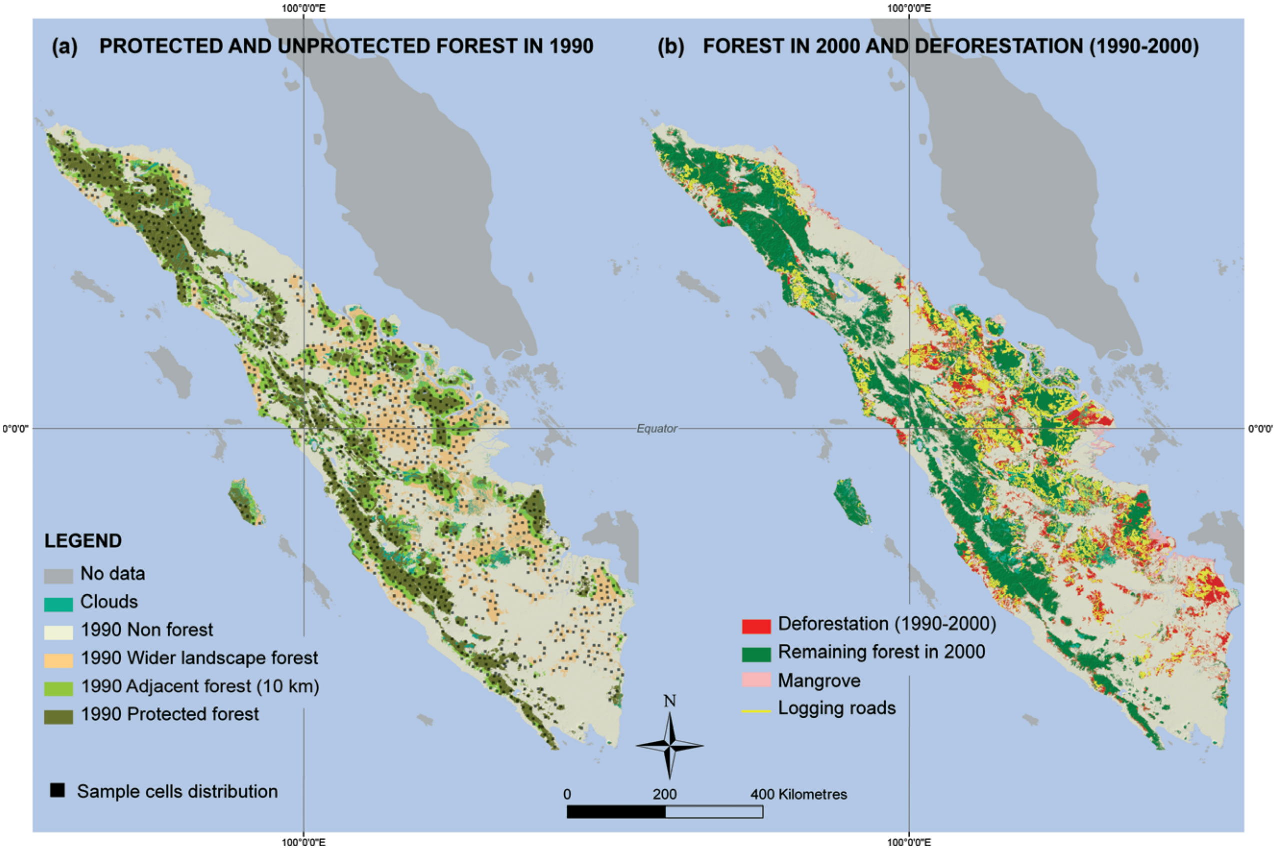

Figure 1 (a) Protected and unprotected forests in 1990 for the main island of Sumatra and the smaller island of Siberut, including adjacent unprotected land lying within 10 km of protected area (PA) boundaries and the wider unprotected landscape, and showing the spatial distribution of the 1264 sample cells (25 km^2). (b) Remaining forests in 2000, deforestation and logging trails occurring during the period 1990–2000 (UTM projection, WGS84). Protected areas (PAs) protecting mangroves or created after 2000 are not shown.

Neighbourhood leakage

The term 'leakage' may refer to how applying PA regulations in one place can displace deforestation somewhere else (Ewers & Rodrigues, 2008). Such leakage can occur over short distances (neighbourhood leakage), for example by pre-emptive clearing, relocations, immigration and development along PA boundaries (Armsworth *et al.*, 2006; Brockington & Igoe, 2006; Oliveira *et al.*, 2007; Wittemyer *et al.*, 2008), and/or over long distances, for example higher timber harvest rates in the wider landscape to compensate for lower rates within PAs, or 'transnational leakage' that may occur as a result of the strengthening of forestry laws and policies in another country (Gan & McCarl, 2007). This study seeks to determine whether 'neighbourhood leakage' has occurred adjacent to Sumatran PAs. We defined adjacent unprotected areas as land lying within 10 km of PA boundaries to compare our results with the simple inside–outside comparisons that have used the same threshold (Bruner *et al.*, 2001; Sanchez-Azofeifa *et al.*, 2003; Nepstad *et al.*, 2006; Wittemyer *et al.*, 2008). We measured neighbourhood leakage by comparing deforestation rates inside PAs and in adjacent unprotected areas with a baseline level (Ewers & Rodrigues, 2008). This baseline would ideally be established by collecting time-series deforestation data at several intervals in time before and after the establishment of PAs, and by comparing temporal fluctuations in deforestation rates for protected forests and for adjacent unprotected forests (Oliveira *et al.*, 2007). Time-series data on deforestation rates were lacking in this island-wide analysis. Therefore, this study chose the wider landscape as the baseline, following the experimental design proposed by Ewers & Rodrigues (2008). Deforestation rates inside PAs and in adjacent unprotected land lying within 10 km of PA boundaries were compared with deforestation rates in the wider landscape. If formal protection has effectively reduced deforestation, deforestation rates inside PAs will be lower than in the sample baseline. Equally, if PAs displaced deforestation to adjacent unprotected areas through neighbourhood leakage, deforestation rates inside adjacent unprotected forests will be higher than in the wider landscape.

Sampling strategy

The study area of 440,000 km^2 was large, so we chose to select sample sites randomly for comparison of deforestation rates. We took this step to: (1) reduce the overall size of the dataset; (2) simplify the statistical analysis; and (3) minimize statistical dependence in the dataset. However, samples can only provide an estimate of the true outcome of the effectiveness of a PA, so a sufficiently large sample size was required to make the analysis statistically meaningful. The trade-off between maximizing the statistical significance of the analysis while minimizing the likelihood of statistical dependence in the dataset was achieved by sampling 1264 cells of 25 km^2 (5 × 5 km) across Sumatra's cloud-free 1990 forest cover map (Fig. 1a). Random cells were selected, but rejected if their centres were within 10 km of a previously chosen cell, to reduce statistical dependence in the dataset. The 1264 sampled cells cumulatively captured 11% of the total 1990 forest cover making up the Sumatra-wide map. Therefore, the results of this study are based upon a sample dataset rather than upon an analysis of all changes in forest cover in Sumatra.

Defining the variables

Response variable

We followed Laurance *et al.* (2002) in using percentage deforestation as the response variable to control for the differing amounts of 1990 forest cover in different cells. The area of deforestation (1990–2000) was extracted for each cell, with values that ranged from 0% to 100% on a continuous scale. However, in contrast to Laurance *et al.* (2002), we did not logarithmically transform our percentages, as transformation can obscure the interpretation of the results while it only slightly improved normality within our dataset.

Key predictor variable

Digital maps of existing PAs created for hydrological (Hutan Lindung in Indonesian) and conservation purposes around 1990 were obtained from the Indonesian Ministry of Forestry at a scale of 1 : 250,000, but were rescaled to 1 : 50,000 wherever such local boundary delineation was available to improve accuracy. Reserves established for conservation purposes included national parks, nature, wildlife and game reserves and recreational parks. Reserves created during the period 1990–2000 were included in the analysis. Reserves created after 2000, or that had no remaining forest cover in 1990, or that only protected mangrove forests were excluded from the analysis ($n = 3$). Hydrological and conservation PAs were combined into one PA category for the analysis of the effectiveness of PAs because local people do not typically distinguish between categories of PAs (Levang *et al.*, 2007). Protected forests, adjacent forests and wider landscape forests existing in 1990 are shown in Fig. 1(a). Cells that fell inside PAs, in the 10 km adjacent area or in the wider landscape were assigned a value of 1 ($n = 463$), 2 ($n = 378$) and 3 ($n = 423$), respectively. This categorical variable was defined as the key predictor variable of our analysis of PA effectiveness.

Confounding variables

Based on the literature on tropical forests and on PA effectiveness, the variables that best determine spatial patterns of deforestation and protection in tropical landscapes are measures of accessibility such as slope and elevation, distance to forest edge, to roads and to logging roads, and measures of agricultural and logging suitability such as soil and forest types (Kaimowitz & Angelsen, 1998; Mas, 2005; Andam *et al.*, 2008). Within Sumatra, the main forest types are determined by elevation (Laumonier, 1997), with high-quality timber mostly found at low elevations. At low elevations, two main forest types prevail: forests that grow on peat or on mineral soils. Small-scale, often migrant landless farmers tend to clear forests based mainly on accessibility as they do not have intimate knowledge of soil types (Benoit *et al.*, 1989). Timber and plantation industries tend to clear lowland forests on both peat and on mineral soils (Uryu *et al.*, 2008). Therefore, slope and elevation, distance to forest edge, distance to roads and distance to logging roads were defined as confounding variables, while soil type was not included.

A digital elevation model from the National Aeronautics and Space Administration's Shuttle Radar Topography Mission (NASA SRTM) was used to generate elevation and slope maps (Rabus *et al.*, 2003). Data for existing road networks were obtained from the Ministry of Forestry. Logging roads, indicative of mechanized logging, were mapped visually as this feature was easily identified on the LANDSAT imagery. The forest edge was extracted from the 1990 forest cover map. However, forest edges formed by small natural clearings in the forest (< 100 ha), by forest lakes and by remote alpine grasslands, such as those found on mountain tops, were not considered as edges. For this purpose, remote alpine grassland areas were identified from Indonesia's Ministry of Forestry Planning Agency (BAPLAN) land-use maps, and our field knowledge. Distance to settlements was not included because of a lack of reliable island-wide information.

Accessibility of the forest was modelled as travel times rather than straight-line distances (Verburg *et al.*, 2004) in order to simulate people on foot walking along the path of least resistance. Travel times are a function of slope, and a GIS algorithm (Spatial Analyst, ARCGIS 9.2) was used to generate a friction map from the slope map. Slope-dependent off-road walking speeds were based on those calculated for a complex agricultural landscape at the forest margin in the Philippines (Verburg *et al.*, 2004), and it was assumed that off-road speeds in forests were similar to those in complex agricultural landscapes. These data were combined to generate travel time maps to roads, logging roads and the forest edge.

The measures of accessibility were extracted at each cell and their mean values calculated. Travel times to roads, logging trails and forest edge were log-transformed to prevent outliers disproportionately influencing the analysis (Laurance *et al.*, 2002). Multicollinearity among variables was also tested to avoid redundancy in the data (Aguilera *et al.*, 2006). One pair of variables, comprising slope and elevation, showed a particularly high degree of intercorrelation (Pearson's $r = 0.92$, $P < 0.01$), so we combined this pair in a single variable using principal components analysis (PCA) (Aguilera *et al.*, 2006). This single PCA variable explained most of the variance observed between slope and elevation (90%), and subsequently replaced the pair as one input in the model. All other pairs were positively correlated, but showed moderate levels of collinearity, below the acceptable level of 0.6 (Green, 1979), and so were not PCA-transformed.

Propensity score matching

Propensity score matching assesses what might have happened had a treatment, in this case protection, not been applied (Rosenbaum & Rubin, 1985; Rubin, 1997; Ferraro & Pattanayak, 2006; Andam *et al.*, 2008; Linkie *et al.*, 2008). A propensity score was defined as the probability of a cell being assigned for protection from a logistic regression. The dependent variable was 0 for unprotected cells ($n = 801$) or 1 for protected cells ($n = 463$). The confounding variables were slope and elevation, distance to forest edge, distance to roads and distance to logging roads. Individual protected or 'treatment' cells were then matched to individual unprotected or 'control' cells that possessed a similar propensity score.

Prior to matching, a factor for political province ($n = 9$) was considered in order to account for important unobserved socio-economic drivers of deforestation that were known to be province specific, including varying governance levels (Smith *et al.*, 2003), migration patterns (Benoit *et al.*, 1989), conflict- and ENSO-related forest fires (Stolle *et al.*, 2003) and industrial plantations (Uryu *et al.*, 2008). The frequency of fires has increased substantially since the 1970s in the eastern provinces of Riau, Jambi and South Sumatra, coinciding with the rapid expansion of large-scale oil palm plantations and logging (Field *et al.*, 2009). Migrations of Javanese farmers have affected the southern provinces of Bengkulu, Lampung and South Sumatra since the 1980s (Benoit *et al.*, 1989). The matching algorithm selected the nearest match for every protected cell from the pool of unprotected cells that fell within the same political province in Sumatra, so that pairs were not too far distant from each other yet possessed similar socio-economic characteristics, thereby reducing the bias caused by unobserved socio-economic drivers of deforestation. Once the algorithm had selected the nearest control cell for a given treatment cell within a given province, this control cell was reconsidered for subsequent matches, allowing the same control cell to be part of several pairs. Once all possible pairs had been formed, pairs containing the same control cell were filtered out: only the pair that showed the smallest difference in propensity scores between treatment and control pairs was retained. All pairs where this difference was > 5% of the propensity score of the treatment cell were excluded to achieve a trade-off between closely matched individual pairs and the size of the sample post-matching. The performance of the propensity score matching was evaluated by investigating whether differences in the confounding variable in the two matched groups of protected and unprotected cells had been eliminated (Rosenbaum & Rubin, 1985). The equality of each covariate histogram distribution across groups was tested using Kolmogorov–Smirnov tests. The equality of each covariate mean across groups was tested using *t*-tests (Rosenbaum & Rubin, 1985).

The propensity score matching was performed three times to pair up: (1) PA cells and wider landscape unprotected cells; (2) adjacent unprotected cells and wider landscape unprotected cells; and (3) PA cells and adjacent unprotected cells. A paired-samples *t*-test computed any differences in the response variable of deforestation rates for each pair, calculated the mean difference and tested whether this mean significantly differed from zero. Like the *t*-test, the Wilcoxon test involved comparisons of differences between measurements but it did not require the assumption of normal data. This analysis was performed in SPSS using a slightly modified version of a matching algorithm developed by Painter (2004).

Table 1 Island-wide losses in forest cover over the main island of Sumatra and the smaller island of Siberut from 1990 to 2000 estimated using LANDSAT satellite imagery. The protected category includes all conservation and hydrological reserves created before the year 2000. The unprotected category includes adjacent land lying within 10 km of protected area boundaries and the wider landscape.

	Whole island	Protected	Unprotected
Forest cover in 1990 (km^2)	205,524	87,115	118,409
Forest cover in 2000 (km^2)	155,446	82,973	72,473
Deforestation (km^2)	50,078	4142	45,936
Deforestation (%)	25.6	5	41

RESULTS

Deforestation patterns across Sumatra and within Sumatran PAs

Nearly half (205,524 km^2) of the 440,000 km^2 study area was covered in natural old-growth forests in 1990 (Fig. 1a). Of these forests, 5% (9982 km^2) were obscured by clouds in 2000 and were discarded in subsequent analyses. By 2000, the overall size of the forest had decreased by 50,078 km^2, representing a 25.6% loss (2.56% $year^{-1}$) in forest cover (Table 1, Fig. 1b). By the year 2000, an estimated 49,020 km of logging roads had been carved out within the forested region, indicating extensive forest degradation (Fig. 1b). In 1990, the combined area of conservation and hydrological PAs contained about 42% (87,115 km^2) of Sumatra's old-growth forests. In the subsequent 10-year period, unprotected areas (including adjacent areas and the wider landscape) had reduced by 45,936 km^2, representing a 41% loss (4.1% $year^{-1}$) in forest cover. By contrast, forests in PAs had shrunk by 4142 km^2, representing

Table 2 Results of the logistic regression model where presence or absence of protection in Sumatra and Siberut islands is the dependent variable. The model outputs the probability of a cell's likelihood to be protected, defined as a propensity score.

Variables	Coefficients	SE	Chi-square	*P*-value
Constant	−11.031	0.747	218.171	0.000
1990 forest cover*	0.001	0.000	27.757	0.000
Slope and elevation†	0.501	0.126	15.724	0.000
Forest edge‡	0.880	0.294	8.973	0.003
Roads‡	1.431	0.256	31.121	0.000
Logging roads‡	1.399	0.163	73.598	0.000
R^2 = 0.653				

*1990 Forest cover per cell was added as a dummy confounding variable to ensure that matched cells possessed a similar amount of forest cover in 1990.

†Slope and elevation denotes the log-transformed first principal component of slope and elevation.

‡Forest edge, roads and logging roads are expressed in travel times, and are log-transformed.

a 5% loss (0.5% year^{-1}) in forest cover (Table 1). But, more than 35% of the Sumatran PAs set aside to conserve biodiversity (n = 40) had experienced severe rates of forest loss (> 1% year^{-1}), while 60% had been encroached by mechanized logging operations (Appendix S2).

Propensity score matching

The logistic regression model correctly predicted 86.2% of the original observations, and had an R^2 value of 0.653 and a receiver operating characteristic (ROC) value of 0.919, indicating a very good fit to the model. The model highlighted the critical role of accessibility in predicting the presence or absence of protection in Sumatra (Table 2). Forests located deeper within the forest interior, on steeper slopes, at higher elevations, further away from roads and logging roads were likely to be placed under protection and were assigned a propensity score that tended towards 1. Forests located near the forest edge, on flatter areas of land, at lower elevations, near roads and logging roads were likely to be left unprotected and were assigned a propensity score that tended towards 0.

The propensity score matching selected: (1) 78 pairs for the PA–wider landscape comparison; (2) 158 pairs for the adjacent area–wider landscape comparison; and (3) 123 pairs for the PA–adjacent area comparison. Matched pairs located in remote areas were found to be under-represented compared

Table 3 Results of *t*-tests and Kolmogorov–Smirnov (KS) tests before and after matching for the main island of Sumatra and the smaller island of Siberut during 1990–2000. (a) Wider landscape cells with PA cells; (b) wider landscape cells with adjacent cells (lying with 10 km of PA boundaries); and (c) adjacent cells with PA cells. Before matching, confounding variables are not balanced across groups as all *P*-values are < 0.05. After matching, confounding variables are balanced across groups as nearly all *P*-values are > 0.05.

	Before matching				After matching			
(a)	Wider mean	PA mean	*t*-test *P*-value	KS test *P*-value	Wider mean	PA mean	*t*-test *P*-value	KS test *P*-value
1990 forest cover	1338.210	2197.96	0.000	0.000	1973.47	1803.65	0.066	0.543
Slope and elevation	−0.752	0.725	0.000	0.000	−0.676	−0.585	0.330	0.314
1990 forest edge	1.491	2.443	0.000	0.000	1.847	1.885	0.565	0.807
Roads	2.114	2.676	0.000	0.000	2.409	2.434	0.710	0.997
Logging road	1.929	2.842	0.000	0.000	1.862	1.914	0.544	0.677
n	423	463			78	78		
(b)	Wider mean	Adjacent mean	*t*-test *P*-value	KS test *P*-value	Wider mean	Adjacent mean	*t*-test *P*-value	KS test *P*-value
1990 forest cover	1338.21	1435.48	0.009	0.000	1607.57	1331.72	0.000	0.001
Slope and elevation	−0.752	−0.046	0.000	0.000	−0.665	−0.465	0.002	0.028
1990 forest edge	1.491	1.737	0.000	0.000	1.656	1.596	0.093	0.203
Roads	2.114	2.270	0.000	0.000	2.184	2.266	0.051	0.565
Logging road	1.929	2.219	0.000	0.000	1.8663	1.947	0.187	0.393
n	423	378			158	158		
(c)	Adjacent mean	PA mean	*t*-test *P*-value	KS test *P*-value	Adjacent mean	PA mean	*t*-test *P*-value	KS test *P*-value
1990 forest cover	1435.48	2197.96	0.000	0.000	1923.65	1932.61	0.899	0.957
Slope and elevation	−0.046	0.725	0.000	0.000	−0.075	−0.0609	0.855	0.811
1990 forest edge	1.737	2.443	0.000	0.000	2.034	2.041	0.859	0.957
Roads	2.270	2.676	0.000	0.000	2.496	2.463	0.439	0.958
Logging road	2.219	2.842	0.000	0.000	2.252	2.271	0.756	0.957
n	378	463			123	123		

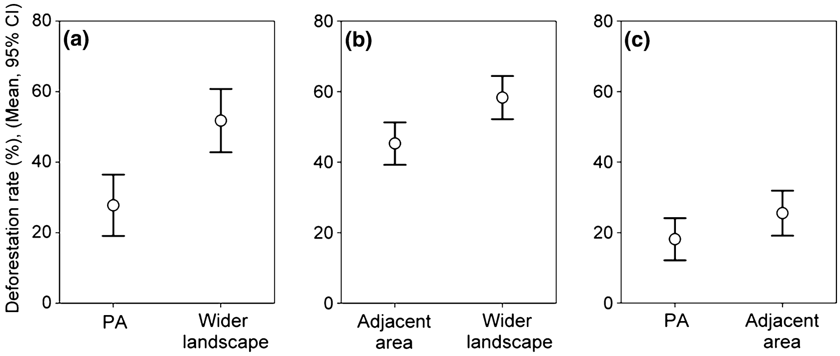

Figure 2 Comparison of deforestation rates inside protected areas (PAs) and in adjacent land lying within 10 km of PA boundaries with deforestation rates in the wider landscape for the main island of Sumatra and the smaller island of Siberut during 1990–2000. Error bars of mean percentage deforestation rates and confidence intervals (CI) after matching for: (a) PA–wider landscape pairs (n = 78); (b) adjacent area–wider landscape pairs (n = 158); and (c) PA−adjacent area pairs (n = 123).

Table 4 Comparison of mean differences in percentage deforestation rates and confidence intervals (CI) across three categories of cells for the main island of Sumatra and the smaller island of Siberut during 1990–2000.

	(1) Wider vs. PA	(2) Wider vs. adjacent	(3) Adjacent vs. PAs
(1) Before matching			
Mean difference (%)	58.6	31.4	27.2
(95% CI)	(54.7−62.5)	(26.3−36.6)	(23.1−31.2)
(2) Propensity score matching			
Mean difference (%)	24.0	13.0	7.4
(95% CI)	(13.7−34.3)	(6.0−20.0)	(1.5−13.3)

Column (1): wider landscape vs. within protected areas (PAs).
Column (2): wider landscape vs. adjacent land (lying with 10 km of PA boundaries).
Column (3): adjacent land vs. within PAs.
The original sample before matching is in row (1). The sample with propensity score matching is in row (2). Mean differences are all positive, indicating that deforestation rates are lower within PAs than in the wider landscape or in adjacent areas. Mean differences are all higher before matching than after matching, indicating that not controlling for variables of accessibility overestimates the protection effect of PAs.

with matched pairs sited closer to the forest edge, on flatter slopes, at lower elevations, closer to roads and logging roads because few Sumatran remote areas have been left unprotected, which explains the small sample sizes obtained after matching. Before matching, the means and histogram distributions of the individual confounding variables (each representing a measure of accessibility) presented large differences between treatment and control groups because all the P-values of the independent samples t-test and two-sample Kolmogorov–Smirnov test were highly significant (Table 3). After matching these differences disappeared, because nearly all the post-matching P-values lost significance (Table 3). Therefore, propensity score matching has balanced measures of accessibility across treatment and control groups relatively well. Any remaining difference in deforestation between treatment and control groups could now be attributed solely to the key predictor variable of the protection status.

The mean difference in percentage deforestation between cells in the baseline of the wider landscape (control) and cells inside PAs (treatment) was highly significant ($t = 4.629$, d.f. = 77, $P < 0.001$), and positive (Fig. 2a, Table 4). This result still appeared robust when the means were compared using a Wilcoxon test ($Z = -3.982$, $P < 0.001$).

The mean difference in percentage deforestation between cells in the baseline of wider landscape (control) and cells inside the adjacent 10-km area surrounding PAs (treatment) was also highly significant ($t = 3.682$, d.f. = 157, $P < 0.001$) and positive (Fig. 2b, Table 4). This result still appeared robust when the means were compared using a Wilcoxon test ($Z = -3.609$, $P < 0.001$).

The mean difference in percentage deforestation between cells in the adjacent 10-km unprotected area (control) and cells inside PAs (treatment) was also significant ($t = 2.499$, d.f. = 122, $P = 0.014$) and positive (Fig. 2c, Table 4). This result appeared less robust but still significant when the means were compared using a Wilcoxon test ($Z = -1.967$, $P = 0.049$). On this basis, the propensity score matching shows that rates of deforestation inside PAs and in adjacent unprotected land were significantly lower than those in the baseline of the wider landscape.

DISCUSSION

This study has substantially improved measurements of 'avoided deforestation' within Sumatran PAs: firstly, in developing fine-scale maps of deforestation over an important region where such information had been previously lacking; secondly, in assessing whether Sumatran PAs have displaced deforestation to adjacent unprotected land (neighbourhood leakage); and thirdly, in controlling for the generally remote location in which Sumatran PAs are established.

At first glance, the results presented in this study mirror those of simpler inside–outside comparisons (Bruner *et al.*, 2001; Sanchez-Azofeifa *et al.*, 2003; Naughton-Treves *et al.*, 2005; Nepstad *et al.*, 2006; Chomitz *et al.*, 2007; Nagendra, 2008) in showing that Sumatran PAs had lower rates of deforestation than unprotected areas during the 1990s. However, this study has shown that not controlling for neighbourhood leakage and for the generally remote location in which Sumatran PAs are established overestimates their 'avoided deforestation' (Table 4). Such results concur with those from another study of the effectiveness of PAs in Costa Rica (Andam *et al.*, 2008): the simple inside–outside comparisons that have been used to date to assess PA effectiveness may have considerably overestimated the protective effect of PAs, in particular where PAs show marked topographic differences between their interior and their immediate surroundings (Mas, 2005).

Furthermore, this study reports the absence of a detrimental neighbourhood leakage effect. Population growth, development projects, pre-emptive clearing and relocations of illegal settlers along the boundaries of PAs appear to have had only a marginal influence on overall deforestation rates around Sumatran PAs. Although pre-emptive clearing has been documented locally in the Peruvian Amazon (Oliveira *et al.*, 2007; Ewers & Rodrigues, 2008), a recent global study concurs with this analysis in showing that there is no evidence for disproportionate population growth near tropical PAs (Joppa *et al.*, 2009). Instead, this study provides evidence for the presence of a beneficial neighbourhood leakage effect. It appears that reducing deforestation inside Sumatran PAs has promoted protection to adjacent unprotected areas. Whether Sumatran PAs extend their conservation influence beyond their boundary may prove controversial, because enhanced law enforcement and ecotourism activities on private lands around PAs are not well developed on the island of Sumatra (Linkie *et al.*, 2008; Gaveau *et al.*, 2009). The unexpected presence of a beneficial leakage effect that conserves forests adjacent to PAs may be explained by an island-wide decreasing population growth effect near Sumatran PAs as human population moves closer to urban centres.

Caution is required if this optimistic PA effectiveness analysis is used to generate conservation policy. Firstly, the results of this study are based upon a sample dataset rather than upon an analysis of all changes in Sumatra. Secondly, this study has not tested whether Sumatran PAs have generated longer-range leakage, for example whether deforestation rates were higher in Sumatra's wider landscape to compensate for lower rates within PAs, or whether transnational leakage might have occurred as a result of the strengthening of forestry laws and policies in another country (Gan & McCarl, 2007). Thirdly, this analysis did not formally include the effects of Sumatra's province-specific variables such as varying governance levels (McCarthy, 2002), migration patterns (Benoit *et al.*, 1989), ENSO-related forest fires (Page *et al.*, 2002; Stolle & Lambin, 2003; Stolle *et al.*, 2003) and the expansion of industrial plantations (Uryu *et al.*, 2008) because of the lack of reliable spatial data on an island-wide scale. A small bias may persist in comparing deforestation between protected and unprotected areas of land, but it has been reduced by matching protected and unprotected cells within the limits of political province boundaries. Therefore, the main conclusions of this analysis of the effectiveness of PAs should be robust.

The real question for policy makers is not whether tropical PAs have lower rates of deforestation than unprotected areas, but rather whether the long-term viability of tropical forests has been secured by establishing PAs. In proportional terms, Sumatran forests reduced at an average rate of 2.56% $year^{-1}$ during 1990–2000, five times faster than the rest of the world's humid tropical forests (Achard *et al.*, 2002). In addition, > 35% of Sumatra's 40 PAs set aside to conserve biodiversity (national parks, nature, wildlife and game reserves and recreational parks) have experienced severe rates of forest loss (> 1% $year^{-1}$) during 1990–2000, while 60% have been encroached upon by logging trails indicating extensive forest degradation (Appendix S2). Sumatran PAs have not halted deforestation within their boundaries and are becoming increasingly isolated ecologically, a pattern paralleled throughout the tropics (Curran *et al.*, 2004; DeFries *et al.*, 2005; Southworth *et al.*, 2006). Sumatran PAs follow the general pattern in allocation seen elsewhere, in that lowland areas, where biological diversity is generally the highest (Whitten *et al.*, 1987; Wilson & Peter, 1988), are preferentially left unprotected relative to highland areas (Leader-Williams *et al.*, 1990; Pressey *et al.*, 1993; Joppa *et al.*, 2008). This trend in PA allocation in part reflects the important consideration of conserving watersheds along mountain ranges. However, it also reflects a lack of support for forest conservation among political elites when faced with opportunities to engage in more profitable extractive forest industries in lowland forests (Curran *et al.*, 2004; Levang *et al.*, 2007; Fitzherbert *et al.*, 2008). Over the last 30–40 years the high economic returns of Sumatra's lowland forest products and agricultural resources have led to consequent government and corporate investments in large-scale forest conversion to plantations (Uryu *et al.*, 2008), a pattern paralleled in Brazil (Ewers *et al.*, 2008). During the late 1970s, Indonesian Borneo and Sumatra changed from being highly fire-resistant to highly fire-prone during drought years because of the rapid increases in large-scale plantations and logging (Field *et al.*, 2009). Sumatra's increasing rural population of small-scale migrant farmers has continued to expand more deeply into the forest frontiers where agricultural land remains abundant (Benoit *et al.*, 1989; Angelsen, 1995; Gaveau *et al.*, 2009). Therefore, the long-term viability of Sumatran forests remains open to question.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Appendix S1 Mapping tropical deforestation in Sumatra.
Appendix S2 Statistics for losses in forest cover and logging from 1990 to 2000, shown for each conservation protected area in Sumatra.

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BIOSKETCH

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