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Deforestation leakage undermines conservation value of tropical and subtropical forest protected areas

Scott Alan Ford[1] [iD] | Martin Rudbeck Jepsen[1] | Naomi Kingston[2] | Edward Lewis[2] | Thomas M. Brooks[3,4,5] | Brian MacSharry[6] | Ole Mertz[1]

[1]Section for Geography, Department of Geosciences and Natural Resource Management, University of Copenhagen, Copenhagen, Denmark

[2]UN Environment World Conservation Monitoring Centre, Cambridge, UK

[3]International Union for Conservation of Nature, Gland, Switzerland

[4]Institute of Marine and Antarctic Studies, University of Tasmania, Hobart, Australia

[5]World Agroforestry Center (ICRAF), University of the Philippines Los Baños, Los Baños, Philippines

[6]European Environment Agency, Copenhagen, Denmark

Correspondence
Scott Alan Ford, Section for Geography, Department of Geosciences and Natural Resource Management, University of Copenhagen, Øster Voldgade 10, 1350 Copenhagen, Denmark.
Email: scf@ign.ku.dk

Abstract

Aim: The establishment of protected areas is among the most widespread responses to mitigate species loss. Although protected areas are often assumed to have conservation benefits, negative impacts have also been documented. One potential negative outcome is leakage, whereby protected areas displace land-use activities harmful to conservation into adjacent areas. This can undermine protection by accelerating loss of species or skewing judgements of effectiveness. This study assessed the prevalence of deforestation leakage in a pan-tropical and subtropical selection of 120 protected areas.

Location: Tropical and subtropical forest regions of America, Africa and Asia.

Time period: 2001–2017.

Major taxa studied: Threatened species of amphibians, birds and terrestrial mammals.

Methods: We used Global Forest Change data to assess the average yearly rate of deforestation in protected areas, protected area buffer zones and statistically matched, unprotected control areas. We calculated and compared irreplaceability of habitat for threatened amphibian, bird and terrestrial mammal species between protected areas and buffer zones.

Results: In 55 cases, deforestation rates were higher in buffer zones than in protected and control areas, suggesting a relatively high prevalence of deforestation leakage stemming from protected areas. In 78.2% of documented leakage cases, reduced deforestation in protected areas was not sufficient to offset the amount of deforestation in 10 km buffer zones to a level that would be expected without protection. In 90.9% of leakage cases, the irreplaceability of species in the 10 km buffer zone was greater than that of the protected area, implying a negative impact of leakage on threatened species.

Main conclusions: The results suggest that protected areas are generally effective at preventing deforestation within their jurisdiction; however, leakage patterns can undermine conservation success because buffer zones often contain habitat for threatened species. We recommend accounting for the possibility of leakage when designing new protected areas and networks.

1 | INTRODUCTION

Protected areas (PAs) are a cornerstone of efforts to conserve species and natural resources throughout the world (Chape, Spalding, & Jenkins, 2008). Accordingly, the 196 parties to the Convention on Biological Diversity (CBD) have adopted Aichi Target 11, committing governments to conserve ≥17% of terrestrial and ≥10% of marine areas through site-based conservation strategies by 2020 (CBD, 2010). Additionally, the integration of PAs into the wider landscape through broadly connected networks and ensuring that they are representative of important sites for biodiversity is a targeted outcome of the expansion of PA estates. Preparations to establish a post-2020 biodiversity framework are underway, and even more ambitious targets for protected area coverage, connectivity and representativeness might be forthcoming (CBD, 2018). To meet these objectives, many CBD member parties might look to expand their PA estates in the coming years (UNEP-WCMC, IUCN, & NGS, 2018). Therefore, identification of the factors that lead to successful area-based conservation outcomes and understanding the interaction of PAs with the wider landscape will be crucial as more land becomes designated for conservation purposes.

Several studies have shown that PAs are an effective means of conservation of biodiversity (Bruner, Gullison, Rice, & Da Fonseca, 2001; Coetzee, Gaston, & Chown, 2014; Geldmann et al., 2013; Gray et al., 2016; Nagendra, 2008; Nelson & Chomitz, 2009). However, they can also result in unintended consequences that undermine conservation efforts (Bode, Tulloch, Mills, Venter, & Ando, 2015; Pfeifer et al., 2012; Renwick, Bode, & Venter, 2015; Visconti et al., 2019). One such consequence is leakage, which occurs when land uses harmful to conservation efforts (e.g., deforestation) are displaced to areas beyond the administrative boundaries of the PA (Ewers & Rodrigues, 2008). When this displacement occurs directly adjacent to the PA, it can alter the composition and structure of vegetation in the PA buffer zone, potentially limiting the ranges and dispersal capabilities of organisms living within the PA and disrupting other ecological functions (DeFries, Karanth, & Pareeth, 2010; Haddad et al., 2015; Hansen & DeFries, 2007). The severity of the impacts of leakage on biodiversity will depend on the biodiversity value (measured, for example, as irreplaceability; Pressey et al. 1993) of the buffer zone compared with the PA. Cases where irreplaceability is lower in the buffer zone are consistent with positive net biodiversity impacts of PAs, but such impact will be minimal where the irreplaceability in the buffer zone and the PA is the same and negative if the irreplaceability of the buffer zone is greater than that of the PA. Leakage in situations where buffer zone irreplaceability is high can reduce species richness and ecosystem function in landscapes surrounding PAs, with feedbacks to species and ecosystem function within PAs. Furthermore, leakage patterns can undermine assessments on the effectiveness of PAs, because success is often determined by comparing ecosystem health indicators between the PA and its immediate surroundings (Ewers & Rodrigues, 2008; Joppa & Pfaff, 2010). The importance of PA buffer zones is recognized by the inclusion of Article 8e in the CBD, which states that parties to the convention should "Promote environmentally sound and sustainable development in areas adjacent to protected areas with a view to furthering protection of these areas" (CBD, 1992). Buffer zones also are recognized by Aichi Target 11, which highlights the importance of integrating site-based conservation measures into the wider landscape (CBD, 2010).

Despite the recognition that areas adjacent to protected areas affect the success of site-based conservation efforts, the global prevalence of leakage remains poorly understood, and few studies have attempted to quantify this phenomenon (Fuller, Ondei, Brook, & Buettel, 2019). Cases of deforestation leakage have been reported in forests of Peru, East Africa and Indonesia (Oliveira et al., 2007; Pfeifer et al., 2012; Poor, Frimpong, Imron, & Kelly, 2019). However, two studies conducted across pan-tropical selections of PAs did not find evidence of widespread deforestation leakage (Fuller et al., 2019; Lui & Coomes, 2016). These studies also offer little insight into the drivers of deforestation in cases where leakage is present, thereby precluding efforts at mitigation.

One explanation for the lack of evidence of deforestation leakage reported in these studies might be the method by which counterfactual deforestation scenarios were established (Fuller et al., 2019). To identify deforestation leakage, the rate of deforestation must be compared among a PA, its buffer zone and an unprotected control (i.e., counterfactual scenario; Ewers & Rodrigues, 2008). The locations of protected areas are non-random. Therefore, any effort to assess the effect of protection on deforestation rates should account for this potential bias, because factors such as elevation and distance to population centres might affect deforestation rates (Joppa & Pfaff, 2009). Statistical matching techniques have been proposed as a method of identifying control areas for calculating counterfactual deforestation scenarios (Ewers & Rodrigues, 2008; Joppa & Pfaff, 2010). This study advances previous efforts by using propensity score matching to quantify and analyse deforestation leakage surrounding tropical and subtropical PAs.

The objective of this study was to identify cases of deforestation leakage in a pan-tropical and subtropical selection of 120 PAs by comparing deforestation rates among PAs, unprotected PA buffer zones and unprotected control areas. We also sought to assess the impact of deforestation leakage from PAs across extensive areas and to assess the main drivers of deforestation in buffer zones of PAs with deforestation leakage.

2 | METHODS

2.1 | Site selection

We used ArcMap v.10.5.1 (Environmental Systems Research Institute, 2019) to select PAs, create buffer zones and create control areas. We randomly selected 120 PAs [America (North and South) = 40, Africa = 40 and Asia = 40] from the October 2018 version of the World Database on Protected Areas (WDPA) (UNEP-WCMC and IUCN, 2018). Our selection included only PAs that were established before 2001 (to accommodate the earliest date of deforestation data), occurred fully within a tropical or sub-tropical forest biome (Olson et al., 2001), were not fully nested within other PAs and for which boundaries were available (i.e., polygons). If the edges of two PAs in the selection were within 20 km, we discarded the selection and made a new selection. This ensured that the 10 km buffer zones of PAs in the selection would not overlap, maintaining the spatial independence of the sample.

We assumed that a pattern of deforestation leakage would displace an equivalent amount of deforestation from the PA to the buffer zone. Accordingly, we sought to analyse deforestation leakage in buffer zones with areas relatively equal to those of their respective PAs. Previous studies analysed leakage within 10 km buffer zones (Fuller et al., 2019; Lui & Coomes, 2016; Poor et al., 2019). However, this buffer radius produces buffer zones with areas similar to that of the PA only when the PA is relatively large (c. 2,000 km^2). Given that most PAs within our sample were considerably smaller than 2,000 km^2 (Supporting Information Table S1), we chose to analyse leakage patterns at four buffer zone levels: 1, 3, 5 and 10 km. We created concentric buffer zones around PAs at these distances and clipped to the extent of tropical and subtropical forest biomes. We then erased areas of buffer zones that overlapped with unselected PAs in the WDPA. This ensured that all buffer zones occurred fully within tropical or subtropical forests and were unprotected.

To identify deforestation leakage, the deforestation rate must be compared between a PA and a control that approximates the area if not protected (i.e., counterfactual scenario; Ewers & Rodrigues, 2008). To create the counterfactual scenario, we overlaid tropical and subtropical forest areas with a 10 km × 10 km grid, with each grid cell representing a potential control area. We then erased cells that overlapped protected areas within the WDPA or 10 km buffer zones of the selected PAs to ensure that control areas were unprotected and were outside the influence of PAs in the sample. We used propensity score matching (PSM) to match PAs with similar but unprotected cells, thereby controlling for covariates in the comparison of deforestation rates (Joppa & Pfaff, 2010). We conducted PSM with the PS Matching extension in SPSS v.25 (IBM, 2019). We performed this analysis separately for Africa, America and Asia. We used distance to the nearest major road, distance to the nearest major population centre, mean elevation, country and terrestrial ecoregion as covariates for PSM, with exact matching on country and terrestrial ecoregion (EEA, 2016; Environmental Systems Research Institute, 2018; Natural Earth, 2018; Olson et al., 2001). We chose these covariates because they can affect rates of

deforestation and have been used in many other statistical matching studies (Andam, Ferraro, Pfaff, Sanchez-Azofeifa, & Robalino, 2008; Mas, 2005; Poor et al., 2019). We used the PS Matching extension to establish callipers that defined a tolerance outside of which matches were excluded from the sample (D'Agostino, 1998). In this manner, we improved the covariate balance by eliminating matches that were relatively dissimilar (Supporting Information Table S3). Each PA was matched with up to five control areas, and for each region we used the minimum callipers that allowed each PA to match at least one control area. This ensured that each PA matched and that the best overall balance was achieved (Andam et al., 2008; D'Agostino, 1998). We set the callipers at 0.19, 0.25 and 0.15 standard deviations of the logit of the propensity scores for America, Africa and Asia, respectively.

2.2 | Deforestation estimates and drivers of leakage patterns

We used the Global Forest Change Dataset (GFCD) 2000–2017 (Hansen et al., 2013) to calculate the average yearly deforestation rate for PAs, buffer zones (1, 3, 5 and 10 km) and control areas. Accordingly, we calculated the average yearly deforestation rate as the area deforested per area of PA, buffer zone or control per year from 2001 to 2017. The GFCD shows tree cover loss at a 30 m × 30 m resolution, with the pixel value corresponding to the first year of detected tree-cover loss. Given that each pixel can contain only one value, the data do not allow for the detection of multiple tree-cover loss events within the same pixel during the time period. We used the GFCD data mask (Hansen et al., 2013) to differentiate terrestrial surfaces from water bodies and considered only terrestrial areas for the calculation of average yearly deforestation rate (Lui & Coomes, 2016). We defined leakage as a higher average yearly deforestation rate in a buffer zone compared with the PA and its matched control area (Ewers & Rodrigues, 2008). To determine whether deforestation rates differed significantly ($p < .05$) among PAs, buffer zones and controls, we performed Kruskal–Wallis and Wilcoxon–Mann–Whitney post hoc tests. We defined significant leakage as an average yearly deforestation rate significantly ($p < .05$) higher in the buffer zone than the PA and control. We assessed potential drivers of leakage patterns with the Drivers of Global Forest Loss (DGFL) dataset, which classifies the main driver of forest loss as commodity production, forestry, shifting agriculture, wildfire or urbanization, at a 10 km resolution with one value per pixel (Curtis, Slay, Harris, Tyukavina, & Hansen, 2018). We determined the prevalent driver of forest loss by counting the number of deforestation pixels within each driver category within buffer zones of PAs that had leakage.

2.3 | Landscape-level effects of leakage

To assess the impact of leakage, we applied the framework of Ewers and Rodrigues (2008). We quantified the area of land deforested across the PA and its 10 km buffer zone over the given time period,

producing a value of deforestation across the "conservation landscape". We compared this value with a counterfactual scenario, in which we calculated the area expected to be deforested in an unprotected landscape of the same size over the same period by extrapolating the area of deforestation in the corresponding control to the size of the combined PA and 10 km buffer zone. By comparing the actual area of deforestation in the conservation landscape with the expected area of deforestation for each case, we classified the pattern as avoided deforestation (actual deforestation < expected deforestation) or enhanced deforestation (actual deforestation > expected deforestation).

To assess further the potential impacts of leakage on landscape-level conservation efforts, we calculated and compared irreplaceability between PAs and their respective 10 km buffer zones. The irreplaceability metric represents the importance of a site for meeting conservation objectives for a given species or set of species (Pressey et al., 1993). A high value of irreplaceability indicates that if a site is lost, the likelihood that another site will replace its conservation value for the given species is low. We calculated irreplaceability following the methods of Le Saout et al. (2013). This method compares the extent of a species' range within the site of interest with its total range, with a higher value of irreplaceability signifying that a site contains a higher percentage of the species' total range. The irreplaceability metrics corresponding to each species that occupies a given site are summed to calculate an overall irreplaceability value for the site. In this study, we considered irreplaceability for threatened species (i.e., IUCN Red List categories Critically Endangered, Endangered and Vulnerable; IUCN, 2012) of birds, terrestrial mammals and amphibians. These taxonomic groups have been assessed comprehensively with IUCN Red List criteria. We acquired range maps from the IUCN Red List for terrestrial mammals and amphibians (IUCN, 2019) and from BirdLife International for birds (BirdLife, 2018). The spatial accuracy of these range maps varies with the extent of the range of each species. Therefore, false-positive errors are inevitable (Rodrigues, 2011) but tend to be smaller for the highly irreplaceable species (i.e., those with a small range), which drive the results.

3 | RESULTS

3.1 | Deforestation patterns

The random selection of PAs across the pan-tropical and subtropical region created a sample of 120 PAs from the 6,679 PAs that met our criteria. These selected PAs were located in 43 countries, encompassed all six IUCN management categories and had areas ranging from 2 to 14,034 km^2 (Figure 1). Propensity score matching techniques achieved good balance for all covariates (standard mean difference < 0.25; Figure 2), indicating that the selection of controls to establish the counterfactual scenario was robust (Thoemmes, 2012). Of the 120 PAs analysed, there were 55 in which the average yearly rate of deforestation was higher in buffer zones than within the PA and its unprotected matched controls (Table 1). The prevalence of leakage was similar for the three regions considered, with leakage in 19, 18 and 18 PAs in America, Africa and Asia, respectively. Of these 55 cases, 18 were significant. The leakage patterns were significant for five, six and seven PAs in America, Africa and Asia, respectively. Leakage was not clearly correlated with IUCN management category, mean elevation, size, age, distance to nearest major road or distance to nearest major population centre (Figure 3).

Across the selected group, average ($\pm SE$) yearly rate of deforestation was significantly lower within PAs (0.25 ± 0.04%) than in all buffer-zone levels (1 km, 0.39 ± 0.05%; 3 km, 0.42 ± 0.05%; 5 km, 0.44 ± 0.06%; and 10 km, 0.43 ± 0.06%) and unprotected controls (0.44 ± 0.05%). The average yearly rate of deforestation was higher in unprotected controls than in 1, 3 and 10 km buffer zones, but equal to that in 5 km buffer zones. Protected areas in America had higher average yearly rates of deforestation at all buffer zone levels than unprotected controls, whereas PAs in Africa and Asia had lower average yearly rates of deforestation at all buffer zone levels than unprotected controls. The average yearly rate of deforestation within the PA was significantly higher than within unprotected matched controls in 15 PAs across the sample, with five cases each in America, Africa and Asia. The average yearly rate of deforestation was significantly lower in PAs than in matched controls for 70 cases: 25 in America, 23 in Africa and 22 in Asia.

FIGURE 1 Locations of 120 protected areas analysed for deforestation leakage
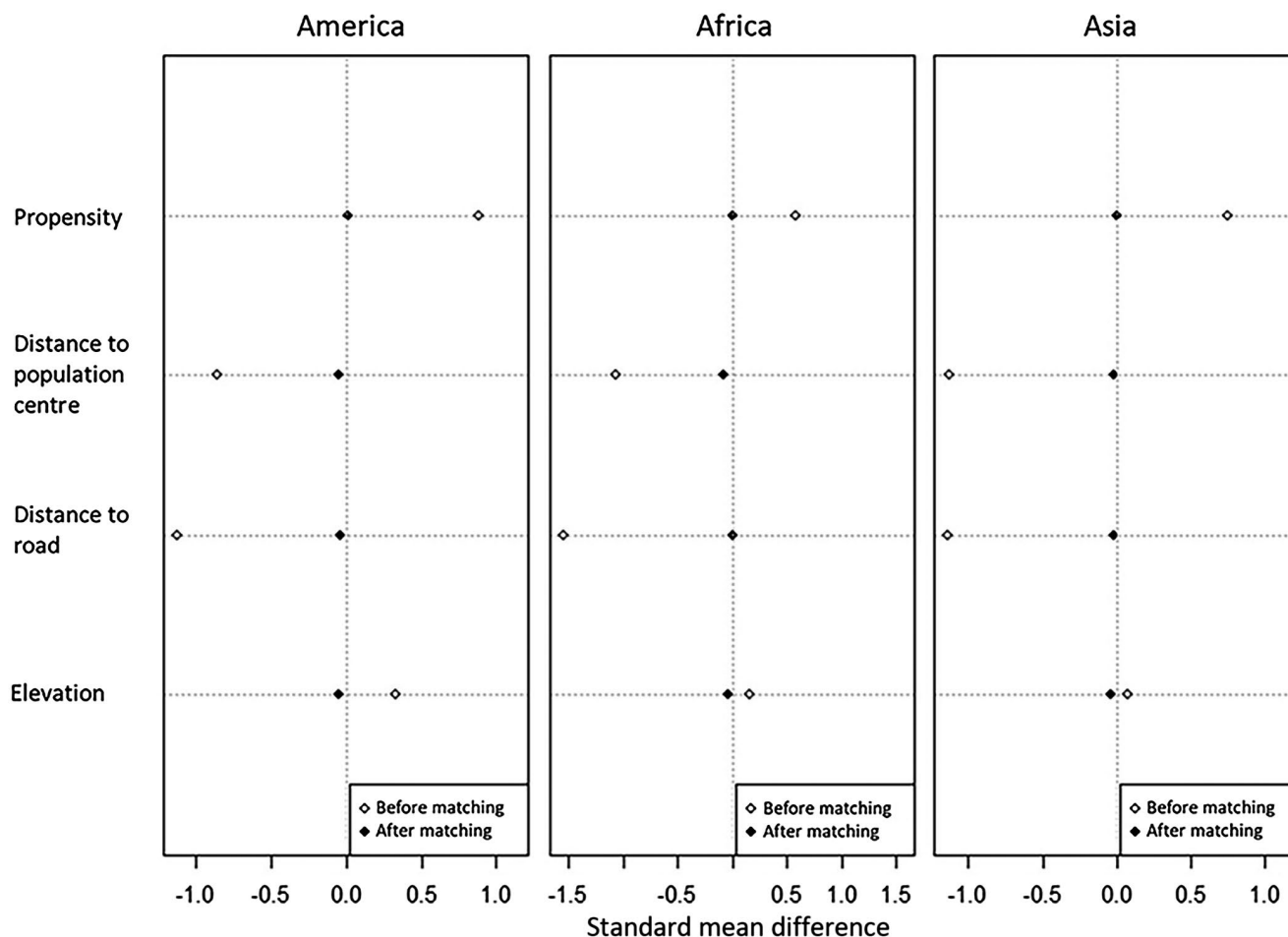
FIGURE 2 The data balance achieved by propensity score matching of protected areas and control cells for America, Africa and Asia. The covariates country and terrestrial ecoregion are not included, because they were matched through exact matching

3.2 | Landscape-scale impacts of leakage

Of the 55 PAs with leakage, 43 (78.2%) resulted in a higher level of deforestation across the conservation landscape compared with the expected level of deforestation without protection, indicating enhanced deforestation (Figure 4). This includes 26 cases in which leakage was present but not significant and 17 cases in which leakage was significant. Protected areas without leakage resulted in less deforestation across the conservation landscape than would be expected without protection in 48 cases (73.8%). Overall, the existence of PAs avoided deforestation in 65 cases (54.2%) and enhanced levels of deforestation in 55 cases (45.8%). This pattern was consistent across all regions, with 21 (52%), 23 (57%) and 21(52%) PAs that avoided deforestation in America, Africa and Asia, respectively.

We calculated irreplaceability for threatened species of terrestrial mammals, amphibians and birds for the 55 PAs exhibiting leakage and their respective 10 km buffer zones (Supporting Information Table S2). Five PAs, three in Africa and two in America, had a higher value of irreplaceability than their 10 km buffer zone (Figure 4). The difference in irreplaceability between buffer zone and PA was >.01 in four of these five cases. Of the 50 cases in which irreplaceability was greater in the buffer zone, the difference in irreplaceability was ≥.01 in six cases.

3.3 | Drivers of deforestation

Among the 55 PAs with deforestation leakage, shifting agriculture was the dominant driver of forest loss in 36 cases (65.5%), followed by commodity production ($n = 10$, 18.2%) and forestry ($n = 9$, 16.4%). The dominant driver of buffer zone forest loss in all 18 African PAs was shifting agriculture. In America, the dominant drivers were shifting agriculture ($n = 13$), commodity production ($n = 4$) and forestry ($n = 2$). In Asia, forestry was the most prevalent driver of buffer zone forest loss ($n = 7$), followed by commodity production ($n = 6$) and shifting agriculture ($n = 5$).

4 | DISCUSSION

4.1 | Global prevalence of deforestation leakage

The results of this study suggest a higher prevalence (46% of PAs sampled) of deforestation leakage from PAs than previously documented. Fuller et al. (2019) found evidence of leakage in 12% of their sample of tropical and subtropical forest PAs, whereas Lui and Coomes (2016) found that 8% of their sample of tropical forest PAs

TABLE 1 Protected areas with deforestation leakage

Protected area	Designation	Country	Annual rate of deforestation (%)			
			Inside PA	Buffer zone	Control	p
Dja	World Heritage Site	Cameroon	0.01[a]	0.26[b]	0.23[b]	<.001
Nouabalé-Ndoki	National Park	Congo	0.00[a]	0.03[b]	0.01[c]	<.001*
Azagny	National Park	Cote D'Ivoire	0.10[a]	1.25[b]	0.95[c]	<.001*
Banco	National Park	Cote D'Ivoire	0.21[a]	0.87[b]	0.50[b]	.002
Moukalaba Dougoua	Faunal Reserve	Gabon	0.03[a]	0.26[b]	0.08[b]	<.001
Assin-Attandanso	Game Production Reserve	Ghana	0.17[a]	0.90[b]	0.60[b]	<.001
Milo	Classified Forest	Guinea	0.21[a]	0.22[a]	0.12[a]	.309
Arabuko Sokoke	National Park	Kenya	0.00[a]	0.24[b]	0.12[b]	<.001
Bahati	Forest Reserve	Kenya	0.58[a]	0.59[a]	0.02[b]	<.001
Kapkanyar	Forest Reserve	Kenya	0.08[a]	0.40[b]	0.07[a]	.001*
Ol Arabel	Forest Reserve	Kenya	0.35[a]	0.57[a]	0.34[a]	.521
Shimba Hills	National Reserve	Kenya	0.17[a]	0.61[b]	0.16[a]	<.001*
Witu	Forest Reserve	Kenya	0.21[a]	1.25[b]	0.10[a]	<.001*
Taylor Creek	Forest Reserve	Nigeria	0.05[a]	0.20[b]	0.19[b]	<.001
Ruvu	Forest Reserve	Tanzania	0.14[a]	0.38[b]	0.33[b]	.003
Assoukoko	Forest Reserve	Togo	0.24a	0.38[a]	0.21[a]	.089
Echuya	Forest Reserve	Uganda	0.08[a]	0.11[a]	0.06[a]	.163
North Rwenzori	Forest Reserve	Uganda	0.08[a]	0.16[b]	0.05[b]	<.001*
Mango Creek 1	Forest Reserve	Belize	0.18[a]	0.61[b]	0.53[b]	<.001
Alto Rio Guamá	Indigenous Area	Brazil	1.35[a]	1.99[b]	1.90[b]	<.001
Apa Jundiaí	Environmental Protection Area	Brazil	0.46[a]	0.52[a]	0.05[b]	<.001
Caetetus	Ecological Station	Brazil	0.01[a]	0.25[b]	0.11[b]	<.001
Itacaiunas	National Forest	Brazil	0.61[a]	1.87[b]	1.17[ab]	.002
Jaci-Paraná	Extractive Reserve	Brazil	3.10[ab]	3.30[a]	1.41[b]	.001
Rio Preto-Jacundá	Extractive Reserve	Brazil	0.88[a]	2.00[b]	0.63[a]	<.001*
Serra Dos Reis	State Park	Brazil	0.05[a]	3.06[b]	0.49[c]	<.001*
Turvo	State Park	Brazil	0.00[a]	0.11[b]	0.07[b]	<.001
Honda y Calderitas	Forest Reserve	Colombia	0.06[a]	0.20[b]	0.17[b]	<.001
Limoncocha	Biological Reserve	Ecuador	0.08[a]	0.80[b]	0.11[a]	<.001*
Bocas del Polochic	Wildlife Refuge	Guatemala	0.05[a]	0.88[b]	0.65[b]	<.001
Cordillera Alux	Watershed Protection Reserve	Guatemala	0.09[a]	0.41[b]	0.14[a]	<.001*
Kaieteur	National Park	Guyana	0.01[a]	0.04[b]	0.03[b]	<.001
Blue Mountain	Forest Reserve	Jamaica	0.09[a]	0.27[b]	0.18[b]	<.001
Kellits-Camperdown	Forest Reserve	Jamaica	0.50[a]	0.55[a]	0.12[b]	<.001
Barranca del Cupatitzio	National Park	Mexico	0.01[a]	0.14[b]	0.10[b]	<.001
Cerro Mactumatza	Statal Reserve	Mexico	0.07[a]	0.33[b]	0.27[b]	.005
Cutervo	National Park	Peru	0.25[a]	0.32[b]	0.09[a]	.033*
Bhadra	Sanctuary	India	0.04[a]	0.08[a]	0.00[b]	<.001
Pabitora	Sanctuary	India	0.00[a]	0.08[b]	0.02[c]	<.001*
Danau Pulau Besar	Wildlife Reserve	Indonesia	0.05[a]	3.46[b]	3.39[b]	<.001
Nantu	Wildlife Reserve	Indonesia	0.07[a]	0.92[b]	0.40[b]	<.001

TABLE 1 (Continued)

Protected area	Designation	Country	Annual rate of deforestation (%)			
			Inside PA	Buffer zone	Control	*p*
Tinombala	Nature Reserve	Indonesia	0.07[a]	0.67[b]	0.23[c]	<.001[*]
Gunung Lumaku	Protection Forest Reserve	Malaysia	0.14[a]	1.56[b]	1.21[b]	<.001
Mandamai	Protection Forest Reserve	Malaysia	0.06[a]	1.96[b]	1.24[b]	<.001
Ulu Kalumpang	Protection Forest Reserve	Malaysia	0.28[a]	2.71[b]	0.86[a]	<.001[*]
Parsar	Protected Area	Myanmar	0.49[a]	0.85[a]	0.22[b]	<.001
Iomare	Wildlife Management Area	Papua New Guinea	0.05[a]	0.17[b]	0.06[a]	.003[*]
Ilog-Hilabangan	Watershed Forest Reserve	Philippines	0.09[a]	0.16[a]	0.09[a]	.920
Maasin	Watershed Forest Reserve	Philippines	0.07[a]	0.08[a]	0.05[a]	.067
Talavera	Watershed Forest Reserve	Philippines	0.03[a]	0.04[a]	0.03[a]	.691
Vappiah Verugal	Reserved Forest	Sri Lanka	0.16[a]	0.38[a]	0.21[a]	.084
Wedakanda	Reserved Forest	Sri Lanka	0.14[a]	0.28[a]	0.16[a]	.248
Yuli	Wildlife Refuge	Taiwan	0.00[a]	0.01[b]	0.00[a]	<.001[*]
Phu Zang	National Park	Thailand	0.13[a]	0.45[b]	0.19[a]	<.001[*]
Salak Phra	Wildlife Sanctuary	Thailand	0.02[a]	0.20[b]	0.07[c]	<.001[*]

Note: Buffer zone values correspond to the highest annual rate of deforestation documented in a 1, 3, 5 or 10 km buffer zone for each case. Annual rates of deforestation in rows followed by the same letter are not significantly different at $p < .05$.

Abbreviation: PA, protected area.

[*]Significant leakage ($p < .05$).

had leakage. One explanation for this discrepancy might be the use of statistically matched control areas to establish the counterfactual deforestation scenario. Protected areas are often established in locations with de facto protection because of high elevation, steep slopes or remoteness (Joppa, Loarie, & Pimm, 2008; Joppa & Pfaff, 2009; Pfaff, Robalino, Sanchez-Azofeifa, Andam, & Ferraro, 2009). By matching PAs in our sample with similarly situated control areas, we accounted for this potential de facto protection effect. This might result in a lower rate of deforestation in the counterfactual scenario than if controls were established via a large buffer ring around the PA (Joppa & Pfaff, 2010), the dominant method in previous studies (Fuller et al., 2019; Lui & Coomes, 2016). A similar effect was documented by studies estimating PA effectiveness. Studies that derived counterfactual scenarios from statistical matching techniques rather than more traditional inside–outside techniques often indicated that PAs were less likely to curb harmful land uses (Andam et al., 2008; Joppa & Pfaff, 2010).

The number of deforestation leakage cases documented across the PA sample was similar for America, Africa and Asia. At the country level, PAs in Brazil, Kenya, Indonesia and Malaysia had relatively high rates of leakage, suggesting that social and political factors might drive deforestation leakage. However, sample size per country was relatively low. Therefore, it remains speculative whether PAs in these countries are more susceptible to leakage. Comparative studies with larger sample sizes for individual countries would be necessary to assess the hypothesis that PAs within certain countries or social and political contexts are more susceptible to deforestation leakage.

4.2 | Landscape-level effects of leakage

In 12 of 55 documented cases of leakage, PAs curb deforestation within their borders to avoid deforestation across the conservation landscape. Although PAs generally are not established with the goal of reducing harmful land uses in areas outside their jurisdiction, overall conservation success might still depend on activities beyond PA borders (DeFries et al., 2010; Hansen & DeFries, 2007; Laurance et al., 2012). This might be the case especially where PAs are small, isolated or inhabited by species with area and resource requirements that are not met fully by the PA itself (Maiorano, Falcucci, & Boitani, 2008; Woodroffe & Ginsberg, 1998). It might be necessary to give particular attention to PAs with these characteristics when considering the potential impact of leakage on overall conservation goals. Accordingly, analysis of leakage within the context of important sites for biodiversity, such as Key Biodiversity Areas, might be relevant to achieving site-based conservation outcomes. Our comparison of the irreplaceability of PAs and their respective buffer zones indicated that in most cases, buffer zones are as important as the PA for threatened species. Two of the 55 PAs with deforestation leakage, Azagny National Park in Côte d'Ivoire and Dja Faunal Reserve in Cameroon, avoided deforestation across the conservation landscape and had a higher value of irreplaceability than their corresponding buffer zones.

4.3 | Drivers of deforestation leakage

Factors we hypothesized to predict deforestation leakage, such as IUCN management category, age, size and isolation of the PA, were
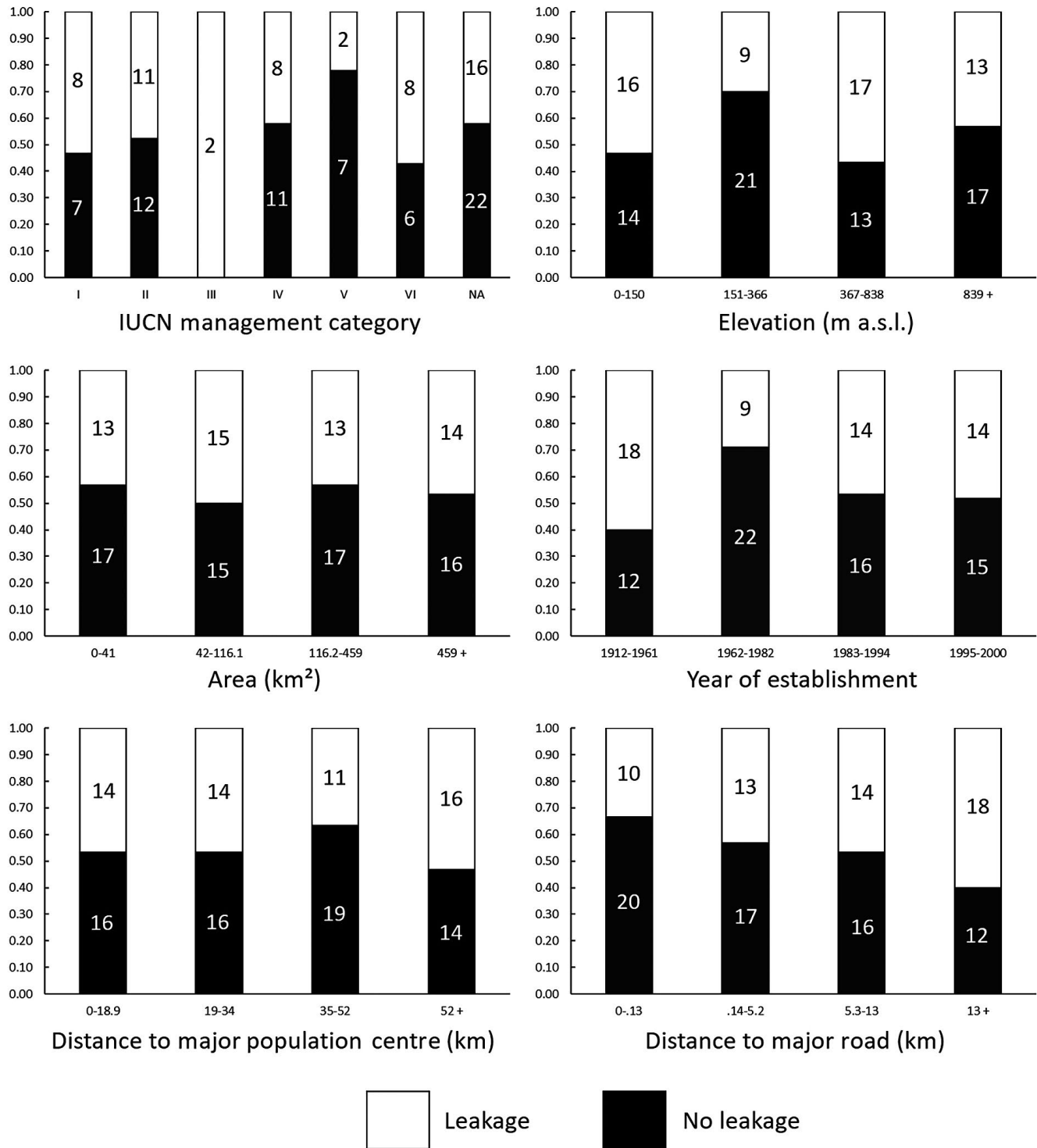
FIGURE 3 Proportional distribution of leakage and non-leakage cases across the following variables: International Union for Conservation of Nature management category, elevation (in metres above sea level), area of protected area (in square kilometres), year of establishment of protected area, distance to major population centre (in kilometres) and distance to major road (in kilometres). Numbers inside bars represent the number of cases per category

not correlated with leakage (Figure 3). This finding is consistent with the patterns reported by Fuller et al. (2019), further suggesting that the factors driving deforestation leakage are case specific and might be controlled by local political, social or economic factors (Lui & Coomes, 2016). Shifting agriculture was the most prevalent driver of forest loss

in buffer zones of PAs with deforestation leakage. However, the DGFL data have relatively low model accuracy in differentiating between shifting agriculture and commodity production in Africa (Curtis et al., 2018). Accordingly, the drivers of leakage for several African PAs might be misclassified. Fifteen of the 18 African PAs with leakage are within
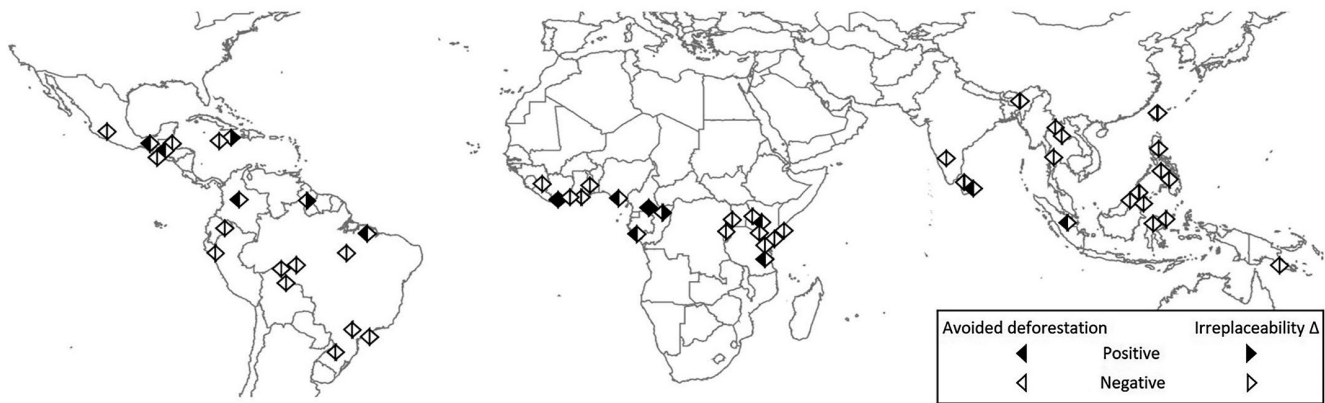
FIGURE 4 Classification of 55 protected areas exhibiting leakage on the basis of avoided deforestation across the conservation landscape and comparison of irreplaceability of protected areas and respective 10 km buffer zones for threatened species of amphibians, birds and terrestrial mammals. A positive avoided deforestation value indicates actual deforestation < expected deforestation across the protected area and 10 km buffer zone from 2001 to 2017. A positive irreplaceability Δ indicates irreplaceability of protected area for threatened species > irreplaceability of 10 km buffer zone for threatened species

landscapes in which shifting agriculture in 2010 was "low" or "very low", whereas the remaining three are within landscapes where shifting agriculture never existed or disappeared long before 2010 (Heinimann et al., 2017). Notwithstanding, commodity production (i.e., permanent conversion of forest to uses such as agriculture, mining or energy infrastructure) was responsible for eight of the nine highest rates of buffer zone deforestation among leakage cases. Although our sample size per country was relatively low, levels of deforestation leakage driven by commodity production were higher in Brazil, Indonesia and Malaysia than in other countries. This corresponds to the general deforestation trends within these countries given that their production economies have largely been driven by deforestation for the expansion of cattle and soy production in Brazil (Alix-Garcia & Gibbs, 2017; Ometto, Aguiar, & Martinelli, 2011) and oil palm in Indonesia and Malaysia (Wicke, Sikkema, Dornburg, & Faaij, 2011). The DGFL data provide a general overview of the possible drivers of deforestation leakage. However, case studies are necessary to pinpoint drivers and why they develop in certain social and political contexts. The synthesis of case studies by van Vliet et al (2012), for example, showed that there are often multiple drivers in each study site. Moreover, whereas shifting agriculture might have been an original and slow driver of change, other drivers, such as commodity agriculture, are leading to rapid replacement of both forests and low-intensity agriculture (van Vliet et al., 2012; Vongvisouk, Broegaard, Mertz, & Thongmanivong, 2016).

Our ability to assess whether drivers of deforestation leakage pre-date PA establishment was limited. The standard theory, and the term leakage itself, suggest that land uses driving leakage occur at the site before the creation of the PA (Ewers & Rodrigues, 2008). In this manner, the legal mechanism of restricting land uses within a geographical zone causes them to shift to the nearest acceptable alternative. Oliveira et al. (2007) documented this spatial and temporal pattern in newly created forest concessions in Peru. However, it remains difficult to assess whether this is the predominant mode through which leakage develops. Accurate data on land use before and after the creation of a PA are not available in many cases (Ewers

& Rodrigues, 2008). These factors confound attempts to document leakage that develops according to the theorized spatial and temporal pattern. Understanding whether leakage patterns stem predominantly from activities that are present before PA establishment or vice versa is important for guiding the decision-making process for new PAs and land uses beyond PAs. For example, if leakage often results from activities that were present before PA establishment, then site selection or financial compensation mechanisms to avoid leakage of these activities into buffer zones becomes important (Bode et al., 2015; Renwick et al., 2015). Conversely, if leakage patterns stem predominantly from activities that begin after PA establishment, then zoning laws pre-emptively restricting some types of development in PA buffer zones might be relevant.

4.4 | Implications for site-based conservation

Our results suggest a higher prevalence of deforestation leakage in tropical forest PAs compared with previous studies. However, it is also apparent that the majority of PAs curb deforestation within their jurisdiction, because 58% of the cases indicated that PAs significantly reduce deforestation within their boundaries compared with what would be expected without protection (Supporting Information Table S1). Additionally, in 40% of the PAs, deforestation was less than would be expected without protection across the conservation landscape, suggesting a blocking effect whereby the buffer zone receives a positive spillover of protection from the PA (Garcia, 2015). This phenomenon has been documented previously (Fuller et al., 2019; Gaveau et al., 2009; Lui & Coomes, 2016). Additionally, PAs considered "ineffective" within our sample are not necessarily ineffective at achieving their overarching conservation goal. Protected areas are designated for myriad reasons, and some PA designations, such as sustainable forest reserves, can have a rate of deforestation that is higher than the surrounding landscape, but sustainable for the given objective.

Given that PAs are likely to remain an integral conservation strategy, we recommend policy measures for buffer zone protection where leakage is deemed both high and likely to cause a considerable impact on the conservation outcome. Legal mechanisms to limit development in PA buffer zones have been established in some contexts, such as the Natural Protected Areas network in Peru (Solano, 2010) and the Baekdu Daegan Mountain System in South Korea (Miller & Kim, 2010). In other cases, voluntary agreements between PA management authorities and private stakeholders promote activities conducive to conservation within buffer zones (Dudley, 2008). Assessing the efficacy of existing buffer zone protection strategies to provide better guidance for the creation of effective legal frameworks that maximize compliance for buffer zone protection in various contexts might be a point of future study.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are from open access sources and available for download from the following websites: WDPA, https://www.protectedplanet.net/; GFCD, https://earthenginepartners.appspot.com/science-2013-globalforest/download_v1.7.html; and DGFL, http://data.globalforestwatch.org/datasets/tree-cover-loss-by-dominant-driver. Range maps: https://www.iucnredlist.org/resources/spatial-data-download (mammals and amphibians) and http://datazone.birdlife.org/species/requestdis (birds).

ORCID

Scott Alan Ford https://orcid.org/0000-0002-7881-8270

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BIOSKETCH

Scott A. Ford is a geographer and conservation scientist specializing in site-based conservation interventions.

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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