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Title

Reduced deforestation and degradation in Indigenous Lands pan-tropically

Author list

Jocelyne S. Sze[1*]; L. Roman Carrasco[2]; Dylan Childs[1]; David P. Edwards[1*]

Affiliations

[1]Ecology and Evolutionary Biology, School of Biosciences, The University of Sheffield, Sheffield, UK

[2]Department of Biological Sciences, National University of Singapore, Singapore

*Corresponding authors: jssze1@sheffield.ac.uk, david.edwards@sheffield.ac.uk

Abstract

Area-based protection is the cornerstone of international conservation policy. The contribution of Indigenous Lands (ILs)—areas traditionally owned, managed, used, or occupied by Indigenous Peoples—is increasingly viewed as critical in delivering on international goals. A key question is whether deforestation and degradation is reduced on ILs pan-tropically and their effectiveness relative to Protected Areas (PAs). We estimate deforestation and degradation rates from 2010 to 2018 across 3.4 Mkm^2 ILs, 2 Mkm^2 of PAs, and 1.7 Mkm^2 of overlapped Protected-Indigenous Areas (PIAs) relative to matched counterfactual non-protected areas. Deforestation is reduced in ILs relative to non-protected areas across the tropics, avoiding deforestation comparably to PAs and PIAs except in Africa, where they avoid more. Similarly, degradation is reduced in ILs relative to non-protected areas, broadly performing comparably to PAs and PIAs. Indigenous support

is central to forest conservation plans, underscoring the need for conservation to support their rights and recognise their contributions.

Main text

Despite international commitments to protect forests under the New York Declaration on Forests and United Nations Sustainable Development Goal 15 (Life on Land), tropical deforestation continues unabated, with primary forest loss in 2019 up 2.8% from the previous year[1]. Tropical forests are central in preserving global biodiversity and retain ~55% of global terrestrial carbon stocks[2,3]. They are also essential to the biocultural identities and livelihoods of Indigenous forest peoples[4]. Alongside increasing amounts of forest lost through commodity frontier expansions for agribusiness, particularly in Southeast Asia, Central and South America, and West Africa[5], there is a lack of legal recognition of Indigenous rights and respect for customary tenure in many countries[4]. This has resulted in land grabs, violence and killings of Indigenous and local peoples defending their land and forests[6].

Area-based protection is a cornerstone in biodiversity conservation policy. These encompass both traditional state-managed protected areas and ‘Other Effective area-based Conservation Measures’ (OECMs), as defined by the Convention of Biological Diversity. The addition of OECMs partly reflect increasing acknowledgment of the injustice caused by PA expansions, resulting in human evictions, killings, loss of traditional livelihoods and cultural identity, and increased conflicts[7]. The upcoming post-2020 global biodiversity framework incorporates proposals to increase targets of the terrestrial surface area under protection, with multiple ambitious visions of where, how, and how much to protect (e.g. 30 by 30, Half-Earth, rights-based approaches)[7,8].

The role of Indigenous Lands (ILs), which include areas and territories conserved by Indigenous peoples and local communities (ICCAs; or ‘territories of life’), is increasingly highlighted as key to achieving these targets[9,10]. Although there are concerns that expanding area-based targets will further the injustices experienced by Indigenous Peoples and local communities[7,11], it is also a transformative opportunity[12]. By increasing recognition and support for Indigenous peoples to secure their land rights and tenure, Indigenous peoples may be able to further contribute to conservation, while retaining their autonomy and land management practices[12,13].

ILs carry significant potential to reduce deforestation and degradation and protect biodiversity at resource frontiers. At least 370 million people who self-identify as Indigenous safeguard and manage or have tenure rights to more than a quarter of the Earth's land surface, intersecting with ~40% of protected areas[14] and at least 36% of Intact Forest Landscapes[15]. In Latin America, ILs have reduced deforestation and degradation rates[16–18], often more so than in state-managed protected areas (PAs)[19–21]. For instance, in Bolivia, Brazil, and Colombia, deforestation rates were lower on titled Indigenous territories compared to matched areas outside by 3 to 88% between 2001 and 2013[17]. However, these studies focused on countries where strong Indigenous movements influenced constitutional reforms, resulting in greater governmental and legal recognition and protection of Indigenous rights and land titling[22,23]. In much of Africa and Asia, enhanced recognition and protection is at best nascent or, at worst, governance and law work against Indigenous land rights[4].

A critical knowledge gap for international policy is the effectiveness of ILs in reducing deforestation and degradation pan-tropically, and whether they perform as effectively as PAs. The lack of understanding of the amount of forest loss within ILs pan-tropically limits the capacity of conservation to appropriately support ILs as a central component of emerging policies and goals for area-based protection. Here, we uniquely quantify deforestation and degradation rates in ILs using recently published geospatial data of where Indigenous peoples have land tenure or *de facto* management of the land globally[14]. The data were obtained from cadastral records for state-recognised Indigenous lands, publicly available participatory mapping, models from census data and maps published in scholarly articles.

To estimate deforestation and degradation rates across the tropics, we used two global datasets. One, the European Commission's Joint Research Centre (ECJRC) dataset on forest cover change in Tropical Moist Forest[24], which defines degradation as short-term (less than

2.5 years) disturbance in tree cover canopy, and deforestation as long-term (more than 2.5 years) conversion to non-forest land. Two, the Global Forest Watch[25] tree cover and loss data, which defines deforestation as a stand-replacement disturbance, or a change from a forest to non-forest state.

We compared tropical moist forest loss and degradation from 2010 to 2019 in 3.4 Mkm^2 of ILs, 2 Mkm^2 of PAs, and 1.7 Mkm^2 of overlapping Protected-Indigenous Areas (PIAs; i.e. PAs on Indigenous lands) relative to matched non-protected areas (Figure 1). We included PAs of all management categories that were designated, inscribed, or established by 2010 and masked out known areas of tree plantations from our analysis (see Methods). Using statistical matching and regression adjustment with the matched samples, we obtain robust estimates of deforestation and degradation under the different protection types[26], whilst accounting for confounding factors including location and accessibility, that affect deforestation and PA establishment and can overestimate intervention effectiveness[19,27].

We conducted the analysis at the 1 km pixel-level for the Americas (33 countries), Africa (26), and Asia-Pacific (23), controlling for country-level effects (see Methods). The Americas have the largest area of tropical moist forests in our study (10.3 Mkm^2), followed by Asia-Pacific (4.9 Mkm^2) and Africa (3 Mkm^2) (Figure 1; Supplementary Table 1). The largest area of PAs and PIAs are in the Americas (1.5 Mkm^2 and 1.4 Mkm^2 respectively), while Asia-Pacific has the largest area of ILs (1.9 Mkm^2).

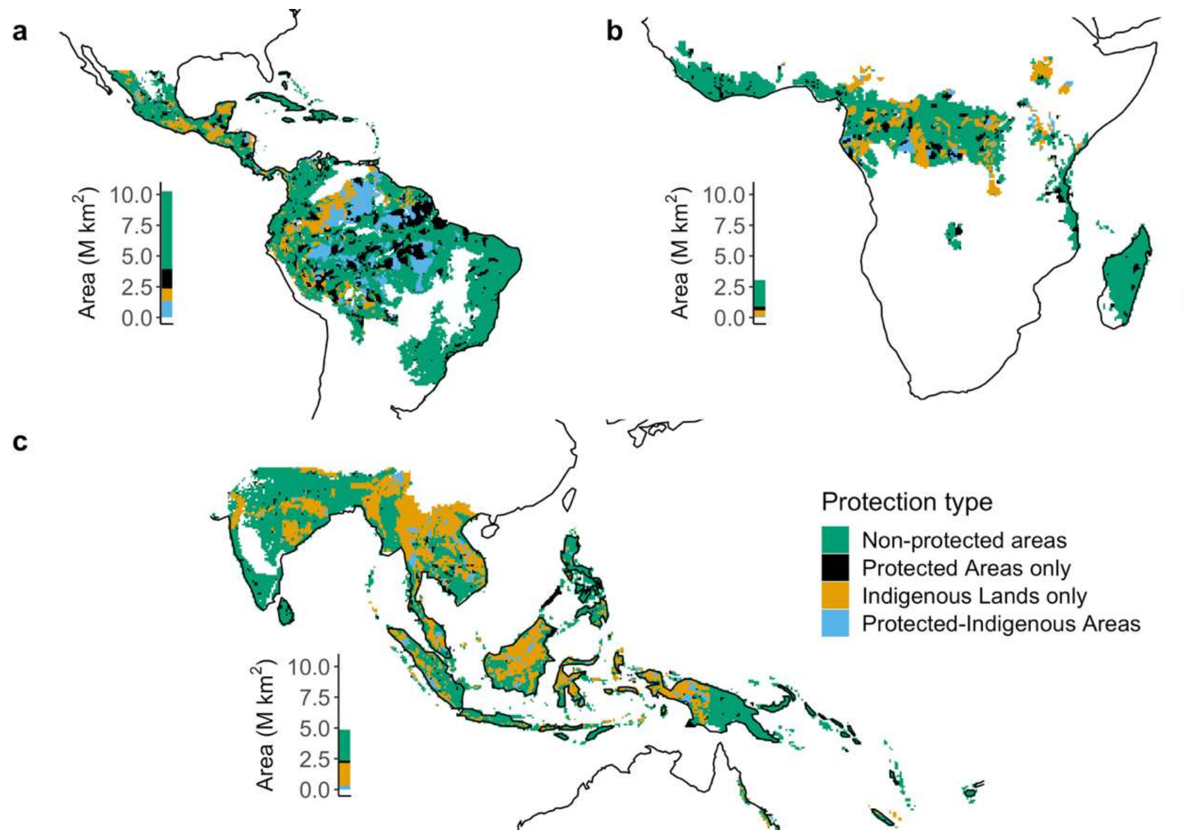

Figure 1. Indicative map of the different protection types across tropical moist forests within our analysis, coarsened to 30 km resolution where each pixel represents the dominant type. The resolution is intentionally coarsened so that boundaries are imprecise, as boundaries of Indigenous territories are often under dispute. A, the Americas. B, Africa. C, Asia-Pacific. See Supplementary Figure 1 for a map of the matched data.

Results

We found that across the tropics, deforestation and degradation rates in ILs were less than in non-protected areas by 16.8-25.9% and 9.1-18.4% respectively (Figure 2; Supplementary Table 2). These rates are broadly comparable to those in PAs, except in Africa, where PAs and PIAs avoid little deforestation and/or degradation. Matching for comparable sites showed the greatest difference in Africa where deforestation and degradation rates were naively overestimated before matching (translucent lines in Figure 2), while they were naively underestimated before matching in the Americas, and in Asia-Pacific, matching and regression showed that protection had reduced forest loss and degradation relative to non-protected areas.

ECJRC deforestation rates were about half that of GFW, though when we combine data on deforested and degraded pixels, the deforestation rates between ECJRC and GFW are more comparable, indicating that GFW considers a lot of short-term degradation as deforestation. Estimated deforestation rates using GFW and ECJRC combined deforestation and degradation data both show that ILs avoid deforestation, though the GFW data estimates greater avoided deforestation in Africa and the Americas (34.3±1.1% avoided in Africa using GFW data versus 21.2±1.3% with ECJRC data, and 29.6±1.5% avoided in the Americas versus 13.9±1.4% respectively) and less in Asia-Pacific (7.5±2% avoided using GFW data versus 16.9±0.7% with ECJRC data).

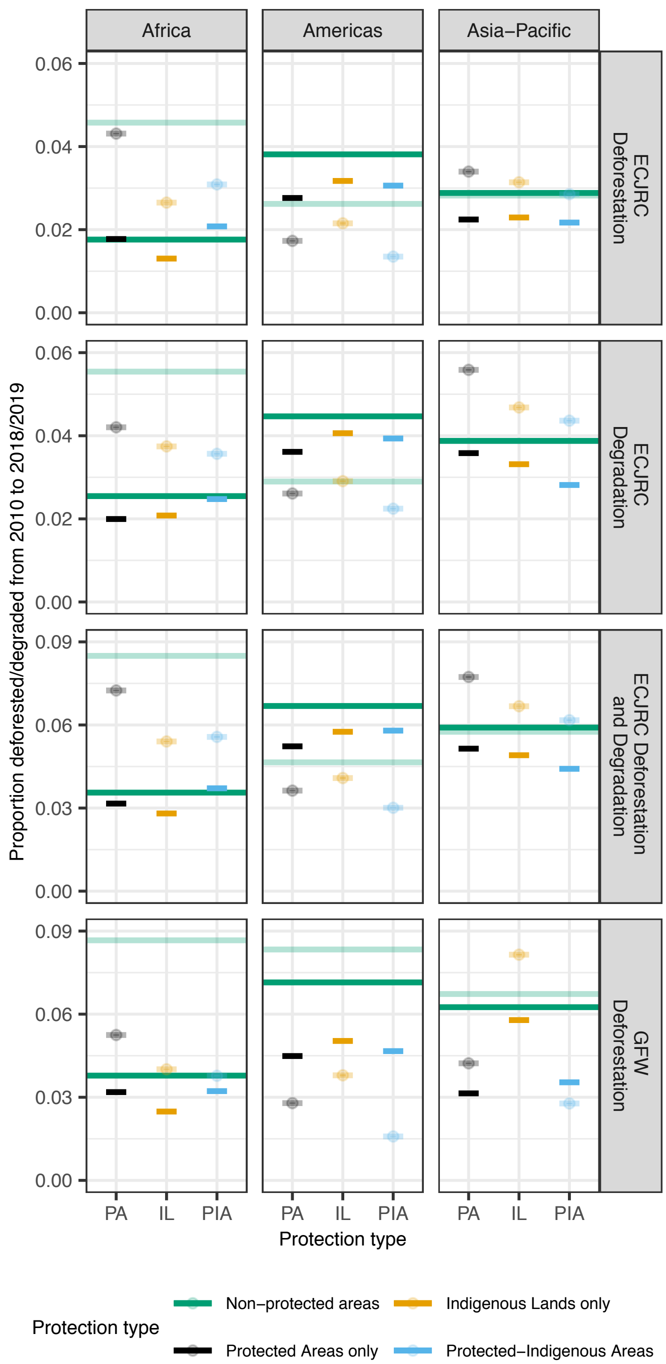

Figure 2. Mean estimated deforestation rates from 2010 to 2019 (or 2018 for GFW data) predicted from GAMM regional models of protection types for ECJRC deforestation rates, degradation rates, combined deforestation and degradation rates, and GFW deforestation rates across tropical moist forest extents, before matching (in translucent colours) and after regression (in solid colours). Vertical lines show standard errors from calculating mean values, which may not be visible at the plotted scale. Values below the solid green horizontal line represent avoided deforestation relative to non-protected areas.

Forest use in ILs may be more similar to that of multi-use PAs (IUCN management categories V and VI), where local forest use is permitted and deforestation rates are higher than strict PAs[28]. Thus, we repeated our analysis including only multi-use PAs and found that ILs still avoid deforestation (21-34.9% for ECJRC data and 11.8-43.3% for GFW data; Supplementary Figures 2 and 3 respectively; Supplementary Table 2). However, focusing on the ECJRC deforestation data, multi-use PAs and ILs in the Americas avoid comparable amounts of deforestation (28.6±0.3% and 21.0±0.4%, respectively) while in Africa and Asia-Pacific, ILs avoid more deforestation than multi-use PAs (by 24.5 percentage points and 11.9 percentage points, respectively). It can take time to develop effective governance and management of PAs[29]. We thus ran a precautionary analysis for PAs that have been established for at least 10 years before our study period (i.e. before 2001), finding broadly similar results to our main analysis that included all PAs established up to 2010 that ILs have reduced deforestation by 14.8-29.8% (Supplementary Figure 2; Supplementary Table 2). However, PAs and PIAs in Africa have higher deforestation rates than non-protected areas, which is in part due to the reduced area of analysis when PAs established between 2000 and 2010 were omitted, such that overall estimated deforestation rates are lower.

To better understand variation in performance across protection types and regions, we performed two further analyses. First, we overlaid classified drivers of forest loss data from Curtis *et al.*[5] on our matched data (Supplementary Figure 1). This revealed that deforestation attributed to commodities and forestry was highest in Asia-Pacific (60-64% of all deforestation in ILs, PAs and PIAs), while in Africa, this fraction was less than 3% (most deforestation attributed to shifting agriculture) and between 41-53% in the Americas, except in ILs where it was 20% (Supplementary Figure 4).

Second, we ran country-specific matching and regression models for 42 countries; these are countries that have sufficient coverage of both PAs and ILs to conduct matching, which omits most countries in West Africa, Caribbean and Oceania. We calculated the regional mean of deforestation rates in non-protected areas and found that higher-than-regional-average deforestation was concentrated in East Africa, Mesoamerica and Southeast Asia (Figure 3; Supplementary Table 3). For these 42 countries, as an exploratory attempt to examine any effect of Indigenous land recognition on deforestation, we identified countries where state governments have recognised Indigenous land ownership or management according to two databases[30,31]. We visualised avoided deforestation in Indigenous lands for the 28 countries that have some Indigenous recognition and those without, but find little association between recognition and avoided deforestation (Supplementary Figures 7-8). We acknowledge that data on secure land tenure and ownership is limited and not spatially explicit; for example, Ecuador is not listed as having any Indigenous ownership or territory, yet in 2008 Ecuador passed a new constitution that recognises Indigenous peoples' rights and ownership of their ancestral lands (Article 84, Constitution of the Republic of Ecuador 2008). As such, we are cautious in interpreting these results.

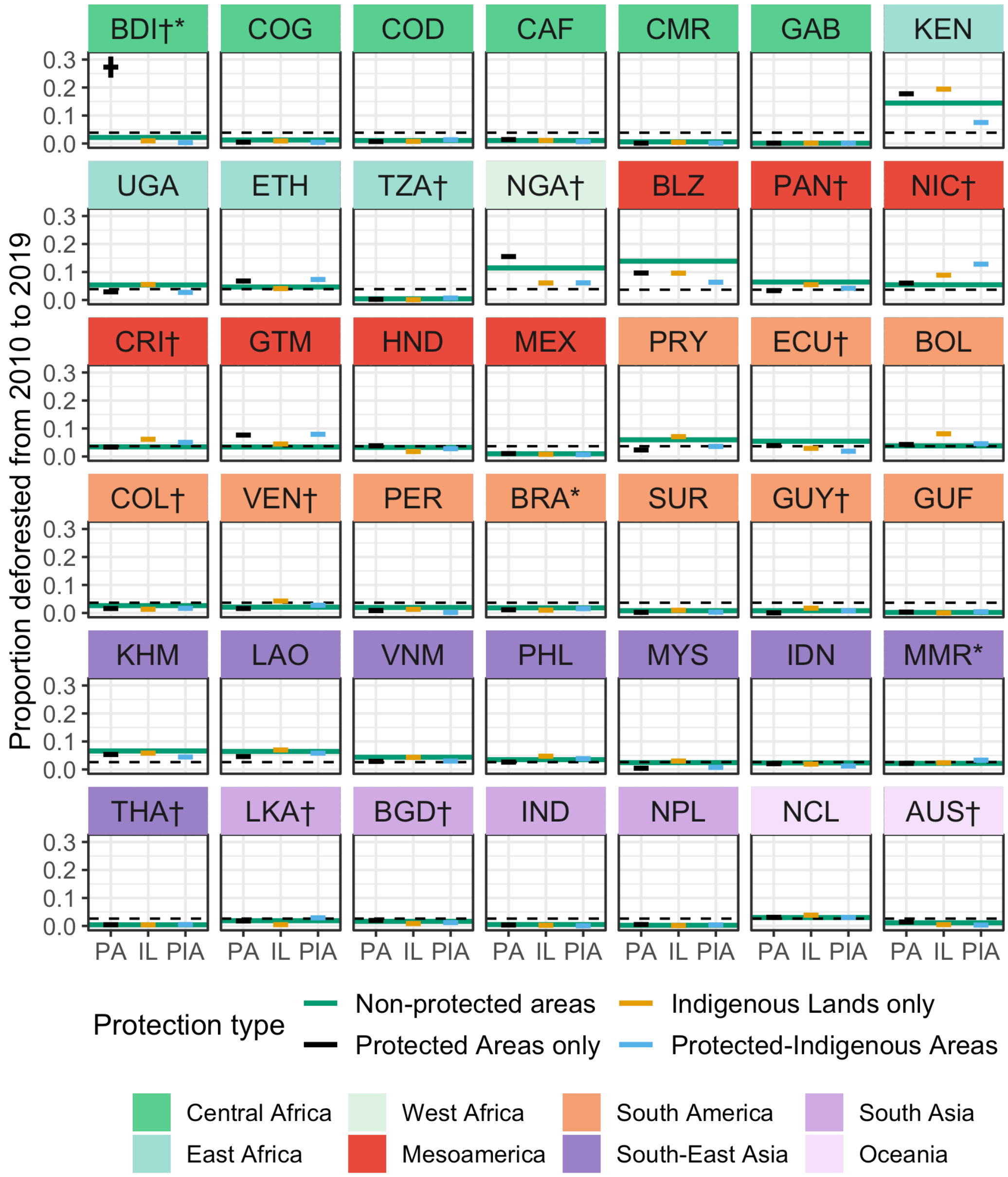

*Figure 3. ECJRC deforestation rates predicted from GAMM models of protection types for each country. Dashed line refers to regional mean deforestation rate of non-protected areas, vertical line shows standard error. Values below the green line represent avoided deforestation relative to non-protected areas. † represent countries for which imbalance remained after matching, * represent countries for which regression residuals did not show any spatial autocorrelation. BDI Burundi; COG Republic of Congo; COD Democratic Republic of the Congo; CAF Central African Republic; CMR Cameroon; GAB Gabon; KEN Kenya; UGA Uganda; ETH Ethiopia; TZA Tanzania; NGA Nigeria; BLZ Belize; PAN Panama; NIC Nicaragua; CRI Costa Rica; GTM Guatemala; HND Honduras; MEX Mexico; PRY Paraguay; ECU Ecuador; BOL Bolivia; COL Colombia; VEN Venezuela; PER Peru; BRA Brazil; SUR Suriname; GUY Guyana; GUF French Guiana; KHM Cambodia; LAO Laos; VNM Vietnam; PHL Philippines; MYS Malaysia; IDN Indonesia; MMR Myanmar; THA Thailand; LKA Sri Lanka;*

BGD Bangladesh; IND India; NPL Nepal; NCL New Caledonia; AUS Australia. See Supplementary Figure 5 for plot with free y-axis scales and Supplementary Figure 6 for results with GFW deforestation data.

Discussion

Drawing on the recent ECJRC tropical moist forest data and widely used GFW global forest change data, our novel analyses reveal that across the tropics, Indigenous Lands have reduced deforestation and degradation rates relative to non-protected areas, despite receiving a much smaller fraction of official conservation funding compared to Protected Areas[7]. While our results reveal deforestation and degradation rates on ILs without identifying causal mechanisms, they reflect the myriad claims by Indigenous peoples who advocate for more recognition on their contributions to conservation and active participation in environmental policy[12,13], and suggests the dovetailing of mainstream conservation efforts and Indigenous protection is a plausible ambition.

Deforestation and degradation on Indigenous Lands

Despite ongoing encroachment on Indigenous Lands[32], we found that ILs have reduced deforestation relative to non-protected areas, performing comparably with PAs and PIAs in the Americas and Asia-Pacific and better in Africa using ECJRC data, but lesser in Asia-Pacific using GFW data (Figure 2). This echoes previous work done in the Amazon[16,17], though we find that across the Americas, PAs do avoid marginally more deforestation than ILs. ILs in Africa avoid more deforestation than PAs or PIAs across all our scenarios. The history of PA establishment in Africa goes back to the colonial era, where the model of fortress conservation is widely applied[33]. Of 34 PAs established in the Congo Basin, 26 resulted in the partial or complete displacement of local and Indigenous communities with no evidence of compensation[34] (e.g., the Twa people and Kahuzi-Biega National Park, Democratic Republic of Congo). In these high-conflict and contentious spaces, it might not be surprising that deforestation rates are higher than in ILs.

While successes in avoided deforestation are a positive outcome, it is encouraging that we found ILs also avoid degradation of the forest canopy. Such degradation is a major source of greenhouse gas emissions[35] and doubles biodiversity losses from deforestation[36]. A myriad of other factors that we were unable to quantify in our analysis also contribute to losses in forest integrity across tropical moist forests[37]. These include alteration of microclimates, proliferation of lianas, and the cascading ecological effects resulting from selective logging, road edges, and reduced populations of large-bodied vertebrates[38,39]. The complexities of monitoring forest degradation using remote sensing not withstanding[40], our finding that short term tree cover loss is reduced relative to non-protected areas and comparably with PAs is encouraging, since 45% of short-term degradation leads to deforestation, particularly in Southeast Africa and Southeast Asia[24]. Selective logging is a key driver of degradation in Southeast Asia and Indigenous communities might be better able than PAs to restrict access to (illegal) logging companies, as some have with oil palm companies[41].

In many Indigenous communities across the world, there exists both formal and informal institutions for governing forest commons and resources[42] and monitoring forest access[43]. The customary forest tenure of Indigenous peoples in Seram island of the Moluccas, Indonesia involve custodians who coordinate forest use, understand the history of forest rights inheritance, and can impose temporary bans on forest access[44]. Similarly, in Ghana, the traditional beliefs and practices of the Ashantis prohibit overexploitation of their forests[45], while the Indigenous peoples in the Xingu, Brazil, mobilise to keep out intruders[46]. Our findings that deforestation and forest degradation are reduced on Indigenous Lands pan-tropically relative to non-protected areas suggest that Indigenous communities and their customary practices do help forest conservation.

Indigenous Rights, Tenure and Land Tenure Security

Indigenous peoples are often invested in protecting their lands from external threats, concurring with our finding that a greater proportion of deforestation is attributed to commodities and forestry within PAs and PIAs than ILs in the Americas and Asia-Pacific (Supplementary Figure 4). However, their ability to steward ecosystems are contingent on state support and protection; even where Indigenous peoples are recognized by state governments and have constitutional rights (e.g. Bolivia, Brazil, Ecuador, Colombia, Republic of the Congo, the Philippines), these rights may not necessarily be implemented or upheld[4]. Indeed, in our exploratory country-level analysis, which utilises the best available information from community-contributed databases that we are aware of, we were unable to discern any patterns between forest protection and countries that recognise Indigenous management and/or ownership (Supplementary Figures 7-8). This may be due to a combination of insufficient or incorrect information, lack of respect or support despite state recognition, or a true lack of association.

Nonetheless, at the regional level, our results reveal that ILs, alongside PAs and PIAs, in the Americas have consistent patterns of reduced deforestation or degradation across our various scenarios (Figures 2 & Supplementary Figures 2-3), which we posit is due to the more advanced legal recognition and protection of Indigenous peoples there compared to other regions, although infringements still occur[4]. Titled Miskitu communities in Bosawas Biosphere Reserve, Nicaragua, were better able to control agricultural expansion threat from mestizo colonisers than were untitled Miskitu communities[47]. However, increasing encroachment and land grabs linked to illicit drug trafficking and the lack of state support have increased deforestation pressure on Indigenous territories[48,49], underscoring the importance of adequate and enforced legal protections, not just titling in name without support.

Extensive global reviews on community management and forest outcomes are increasingly converging on finding that communities with secure land tenure, local autonomy in management, established internal institutions, and supportive national policies have lower deforestation and deliver socio-economic, biodiversity, and climate benefits[50–52]. Although our analysis is unable to account for land tenure security, our results mostly echo previous findings on titling Indigenous territories, where *de jure* titling helped Indigenous peoples slow agricultural expansion and deforestation in their territories. Future research could examine Indigenous recognition, protection, and secure land tenure and refine our work on their impacts on forest protection. However, land tenure systems and their effectiveness are often highly context-dependent and in-depth grounded case studies would be better able to understand these linkages[53]. The process of formalizing land tenure may also bring additional problems[54], such as reducing community autonomy and security[55].

Limitations

There are difficulties in defining Indigeneity at the policy level especially in Africa and Asia-Pacific, and we acknowledge that maps are subjective and partisan[56]. Hence, knowing whether Indigenous peoples are actually present and actively able or willing to conserve their bio-cultural heritage is empirically difficult. In lieu of this, interpreting results remain challenging, particularly in Africa and Asia-Pacific. Further, western-based conservation organisations have often struggled to foster inclusive and equitable participation with Indigenous peoples[57]. Thus, enrolling Indigenous peoples to fulfil internationally set targets raises important ethical and moral concerns; critics argue Indigenous peoples' perspectives and ontologies should be given primacy whilst decentring western ideals[58]. Moving beyond tokenistic inclusion of Indigenous peoples is necessary and potential misalignments in goals of Indigenous communities and conservation will need to be debated.

PAs have spillover effects of either leakage or blockage (i.e., higher or lower deforestation in unprotected adjacent surroundings), and in the Brazilian Amazon, ILs increase leakage while federal PAs increase blockage[59]. However, we have not accounted for spillover effects in our analysis, which would be a refinement for future research. Additionally, we matched on variables that influence deforestation probabilities and biases in locating protected areas, and assumed similar confounders for degradation. We used short-term tree cover loss as a measure of forest degradation, which omits other definitions and measures of forest degradation and losses in forest integrity[37]. Future research could examine how PAs and ILs avoid deforestation where threats are similar (i.e. matching for areas where deforestation is mostly driven by commodities, forestry, wildfires, or shifting agriculture) and consider other industrial pressures on tropical forests. Our overlay of deforestation drivers showed that ILs across the tropics faced less deforestation from commodity and forestry compared to PAs (Supplementary Figure 4), however, given the coarser resolution of the data (~10 km) and difficulties in identifying deforestation drivers from satellite imagery, a more focused analysis would provide further nuance to our understanding of how different protection types help avoid deforestation and degradation, alongside grounded case studies. Further, while forest protection is a key aspect of biodiversity conservation, the human use of wildlife has resulted in local extirpations, affecting tree dispersal and potentially the long-term survival of tropical forests[39].

Conclusion

Protected areas and Indigenous territories are often threatened by the same macroeconomic political forces[60]. Planned investment increases in agriculture, economic growth-inducing infrastructure, and road networks will likely accelerate deforestation and biodiversity loss in the coming decades[61], further increasing pressures both inside and outside protected zones. For instance, high deforestation rates in Cambodia linked to large-scale land acquisitions[62] meant that PA and IL deforestation rates were still higher than the regional average (Figure 3 and Supplementary Figures 5-6), while the expansion of foreign direct investments and large-scale land acquisitions in Africa[63] points towards possible future incursions into Indigenous lands. Strengthening Indigenous Peoples' rights, providing secure tenure and conservationists actively supporting environmental defenders and Indigenous communities will be a vital component of the coordinated action necessary to ensure the survival of tropical forests into the Anthropocene.

Methods

We defined the spatial extent of tropical moist forests as the study area for this analysis, using the data available from ECJRC. See Supplementary Methods for more details. Spatial data cleaning was done in R version 3.6.2, Python 3.7.0, QGIS version 3.4.6 and ArcGIS Desktop 10.7.1 ArcMap, while statistical analyses were done in R version 3.6.3.

Data

Deforestation and degradation from 2010 to 2019 Tropical Moist Forest Subtypes were downloaded from the European Commission Joint Research Centre[24]. We used classes defined as deforestation from 2010 to 2016 and deforestation/degradation from 2016 to 2019, and classes defined as degradation from 2010 to 2016 and deforestation/degradation from 2016 to 2019 to define deforestation and degradation respectively. We additionally combined deforestation and degradation classes to compare with GFW data. To improve computational tractability, we aggregated ~30 m pixels of the original dataset to 1 km, using number of deforested and/or degraded forest pixels (out of maximum of 961 pixels) to fit binomial models of probability of losing a 30 m pixel. This was processed in R environment.

Forest loss from 2010 to 2018 Tree cover 2010[64] and 2011 to 2018 forest loss data (v1.6)[65] were downloaded from Global Forest Watch[25]. We defined pixels as being forested in 2010 using 25% tree canopy cover threshold, as recommended by Ref. [66] to be the threshold which can identify tall woody vegetation unambiguously. Forested pixels that were not lost between 2011 and 2018 were considered to still be forested in 2018. To improve computational tractability, we aggregated 30 m pixels of the original dataset to 1 km, using the number of forested pixels in 2010 and 2018 (out of the maximum 1024 pixels) to fit binomial models of probability of losing a 30 m pixel. This was processed in the Python environment.

World Database of Protected Areas Jan 2020 version was obtained from UNEP-WCMC[67] using the R package wdpar[68] and cleaned according to the recommended protocol, omitting point data due to lack of area information. The IUCN categorises PA based on management objectives; we considered categories I-IV to be strict PAs and categories V and VI to be multi-use PAs. We included all IUCN categories for PAs created before 2011 (including PAs with no information on year of establishment). This resulted in 1740 strict PAs, 939 multi-use PAs and 1276 uncategorised PAs (Supplementary Table 8). We conducted a separate matching and regression analysis with only multi-use PAs, and including only PAs created before 2001 (1279 strict PAs, 587 multi-use PAs and 995 uncategorised PAs), with similar results (Supplementary Table 2).

Indigenous Peoples Lands was obtained from Garnett *et al*.[14] We acknowledge that Indigenous lands were mapped based on publicly available information[14] and that land not mapped as Indigenous are not necessarily non-Indigenous. We also realise that boundaries may be under contestation, and we do not assert their use here as a political statement.

Protected-Indigenous Areas Where boundaries of Indigenous Peoples Lands and PAs overlapped, the spatial intersection of the two layers were considered as Protected-Indigenous Areas (PIAs). PAs that were listed as being 'governed by Indigenous People' within the WDPA database were also considered under this category of protection (467 PAs). The PAs in this PIA layer (established before 2011) consisted of 722 strict PAs, 327 multi-use PAs and 713 uncategorised PAs (note as this layer includes the spatial intersect of WDPAs and IPLs, some of the PAs counted may only be partially represented).

Non-protected areas consists of areas that do not fall under PA or IPL or both, up to January 2020; 1777 PAs created between 2011 and 2019 were excluded. To ensure that pixels were 100% either in a protected category or not protected, we omitted all border pixels from our analyses. As forest loss data from Global Forest Watch include tree plantations, we also masked out known tree plantations[69,70] from our analysis.

Matching covariates Following previous studies (e.g. refs. [19,20,71,72]), we included variables that affect deforestation and assignment of PAs; the nature of ILs being under the ownership and/or management of Indigenous Peoples since before nation states were formed and contemporary political economic factors that affect ILs are such that there are numerous unobservables that cannot be controlled for. Nonetheless, we controlled for confounding variables where possible in both our matching and regression analyses. See Supplementary Table 4.

Baseline forest cover We used tree cover 2010[64] at 25% tree canopy cover threshold to calculate the baseline forest cover in 2010 for matching with PAs established before 2011, and tree cover 2000[73] at 25% tree canopy cover threshold to calculate the baseline forest cover in 2000 for matching with PAs established before 2001.

Slope and elevation data were obtained from Amatulli *et al.*[74] These influence likelihood of deforestation and PA locations[72].

Travel time to nearest populated area with more than 5000 population in 2015 was obtained from Nelson *et al*.[75]. Access to markets and transport hubs influence land-use change decisions.

Distance to roads was calculated from CIESIN SEDAC[76] gROADS dataset in ArcMap using Euclidean distance, as proximity to roads is a major driver of deforestation.

Population density in 2010 from WorldPop[77] was included to control for forest pressure from local human populations.

Countries from GADM[78] were included as an exact matching variable and random effect in regression models to control for country-level factors such as legislation and political-economic situations.

Analysis

All spatial data were gridded to 1 km resolution in EPSG 4326 coordinate reference system and split by IPBES regions (Africa, Americas and Asia-Pacific) for matching and regression. While matching for individual polygons of PAs, ILs and PIAs and comparably sized non-protected areas would have been a refinement of our work that takes protection type sizes into consideration, to protect Indigenous communities from possible unintended consequences, individual IL polygons defining communities are not available. As such, we opted to match at the pixel level. Considering the tropical moist forest extent for PAs established before 2011, this resulted in the following numbers of 1 km pixels for: Africa – 2567650 non-protected, 352365 PA, 554076 IL, and 120799 PIA; Americas – 7579846 non-protected, 1814842 PA, 1222916 IL, and 1583040 PIA; Asia-Pacific – 3033714 non-protected, 203194 PA, 2268545 IL, and 352750 PIA (Supplementary Table 5). For countries where all three protection types were present (42 countries), we conducted additional analyses matching and regressing at the country-level.

Statistical matching We used propensity score matching to select counterfactuals; the propensity score is the probability of receiving the intervention given the baseline covariates, and control observations are matched to treatment observations with the closest propensity score. This reduces imbalance/bias between the covariates, measured using the standardised difference in means (SMD) of the covariates. Matching is considered to have improved balance if the SMD is less than 0.25[26].

Matching was conducted separately for each of the protection type for each IPBES region. To keep analysis tractable, we took samples of the data to eventually yield ~100,000 pixels post-matching (out of 2.7 to 9.4 million pixels for each type-region dataset); 100,000 was the maximum eventual sample size to be able to carry out the matching process within a reasonable time period. We used the MatchIt package in R[79] on the samples, with the default logit method and 1:1 nearest neighbour match without replacement and caliper size of 0.25 of the standard deviation of the estimated propensity score to ensure good matches. We included all numeric covariates (slope, elevation, population density, travel time, distance to roads, and forest area in 2000 or 2010), with exact matching for country. We took 5 separate samples, yielding a total of 3.3 million pixels, and checked that balance was improved from the matching (Supplementary Table 5); only ILs in Americas and PIAs in Asia-Pacific did not have improved balance after matching. Matching at the country level did not reduce imbalance in 20 instances (Supplementary Table 6).

Regression adjustment Matching alone does not completely eliminate imbalance in all covariates across all observations, and the additional regression conducted sought to resolve this. As such, we combined the matched datasets for the different protection types for each region, removing duplicate observations (of non-protected pixels). There were no strong correlations ($r > 0.7$) between the covariates of each region, apart from elevation and slope in the Americas. At the country level, strong correlations were also observed mostly between slope and elevation, which is expected.

We fitted Generalised Additive Mixed Models (GAMMs) for each region using the mgcv package in R[80], including a parametric term for protection type, numeric covariates as cubic regression smoothing splines and geographic coordinates as an additional spline to reduce spatial autocorrelation (with default thin spline smoothing basis). We fit country-level random slopes for 2010 forest area (for PAs established before 2011, or 2000 forest area for PAs established before 2001), an interaction term between protection type and country, and a random intercept for country. We used the bam function for large datasets, default fREML method, binomial family with logit link, with the argument discrete=TRUE to ensure model convergence. We examined autocorrelation plots of model residuals and no pattern was observed. We also fitted models without geographic coordinates and examined model residuals to check if possible confounding factors have been included and find our models robust to endogeneity. We used the models to predict deforestation/degradation rates for each region, holding numeric covariates constant at their means and the longitude and latitude coordinates and countries from the input data. For the country models, we fitted similar models, but used only data from each country and so did not include any country-related random effects. Most model residuals showed some spatial autocorrelation.

Data availability

The data that support the findings of this study are all publicly available online (see Supplementary Table 4 for full source details). The map of Indigenous Peoples Lands can be obtained from the authors upon reasonable request (Garnett *et al.* 2018).

Code availability

Code used for the analysis can be found in the Supplementary Methods section.

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Author contributions

Correspondences and requests for materials should be addressed to JSS and DPE. All authors contributed to the conceptualisation and design of the study. JSS processed and analysed the data, and all authors interpreted the results. JSS led the writing of the manuscript, and all co-authors contributed critically to the drafts.

Competing interests

The authors declare no competing interests.

Figure Legends

Figure 1. Indicative map of the different protection types across tropical moist forests within our analysis, coarsened to 30 km resolution where each pixel represents the dominant type. The resolution is intentionally coarsened so that boundaries are imprecise, as boundaries of Indigenous territories are often under dispute. A, the Americas. B, Africa. C, Asia-Pacific. See Supplementary Figure 1 for a map of the matched data.

Figure 2. Mean estimated deforestation rates from 2010 to 2019 (or 2018 for GFW data) predicted from GAMM regional models of protection types for GFW deforestation rates, combined ECJRC deforestation and degradation rates, and separate ECJRC deforestation and degradation rates across tropical moist forest extents, before matching (in translucent colours) and after regression (in solid colours). Vertical lines show standard errors from calculating mean values, which may not be visible at the plotted scale. Values below the solid green horizontal line represent avoided deforestation relative to non-protected areas.

Figure 3. ECJRC deforestation rates predicted from GAMM models of protection types for each country. Dashed line refers to regional mean deforestation rate of non-protected areas, vertical line shows standard error. Values below the green line represent avoided deforestation relative to non-protected areas. † represent countries for which imbalance remained after matching, * represent countries for which regression residuals did not show any spatial autocorrelation. BDI Burundi; COG Republic of Congo; COD Democratic Republic of the Congo; CAF Central African Republic; CMR Cameroon; GAB Gabon; KEN Kenya; UGA Uganda; ETH Ethiopia; TZA Tanzania; NGA Nigeria; BLZ Belize; PAN Panama; NIC Nicaragua; CRI Costa Rica; GTM Guatemala; HND Honduras; MEX Mexico; PRY Paraguay; ECU Ecuador; BOL Bolivia; COL Colombia; VEN Venezuela; PER Peru; BRA Brazil; SUR Suriname; GUY Guyana; GUF French Guiana; KHM Cambodia; LAO Laos; VNM Vietnam; PHL Philippines; MYS Malaysia; IDN Indonesia; MMR Myanmar; THA Thailand; LKA Sri Lanka; BGD Bangladesh; IND India; NPL Nepal; NCL New Caledonia; AUS Australia. See Supplementary Figure 5 for plot with free y-axis scales and Supplementary Figure 6 for results with GFW deforestation data.

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