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CONTRIBUTED PAPERS

# Rates of tree cover loss in Key Biodiversity Areas on Indigenous Peoples' lands

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**Article impact statement**: Tree cover loss in forest key biodiversity areas is lower on Indigenous lands, but this varies, likely reflecting recognition of their rights.

## Abstract

Indigenous Peoples' lands (IPL) cover at least 38 million km$^2$ (28.1%) of Earth's terrestrial surface. These lands can be important for biodiversity conservation. Around 20.7% of IPL intersect areas protected by government (PAs). Many sites of importance for biodiversity within IPL could make a substantial but hitherto unquantified contribution to global site-based conservation targets. Key Biodiversity Areas (KBAs) represent the largest global network of systematically identified sites of high importance for biodiversity. We assessed the effectiveness of IPL in slowing biodiversity loss inside and outside PAs by quantifying tree cover loss from 2000 to 2019 in KBAs at international and national levels and comparing it with losses at equivalent sites outside mapped IPL. Based on a matched sample of 1-km$^2$ cells in KBAs inside and outside mapped IPL, tree cover loss in KBAs outside PAs was lower inside IPL than outside IPL. By contrast, tree cover loss in KBAs inside PAs was lower outside IPL than inside IPL (although the difference was far smaller). National rates of tree cover loss in KBAs varied greatly in relation to their IPL and PA status. In one half of the 44 countries we examined individually, there was no significant difference in the rate of tree cover loss in KBAs inside and outside mapped IPL. The reasons for this inter-country variation could illuminate the importance of IPL in meeting the Convention on Biological Diversity's ambition of conserving 30% of land by 2030. Critical to this will be coordinated action by governments to strengthen and enforce Indigenous Peoples' rights, secure their collective systems of tenure and governance, and recognize their aspirations for their lands and futures.

**KEYWORDS**
deforestation, forest loss, Indigenous Peoples, protected areas, site-based conservation

Tasas de pérdida de la cobertura arbórea en áreas clave de biodiversidad en suelo indígena

**Resumen:** Las tierras de los pueblos indígenas (TPI) cubren al menos 38 millones de km$^2$ (28.1%) de la superficie del planeta. Estas tierras pueden ser importantes para la conservación de la biodiversidad. Un 20.7% de las TPI se intersecan con áreas protegidas (AP) por el gobierno. Muchos sitos con importancia para la biodiversidad dentro de las TPI podrían contribuir de forma sustancial, pero todavía sin cuantificar, a los objetivos globales de conservación in situ. Las áreas clave para la biodiversidad (ACB) representan la mayor red mundial de sitios con identificación sistemática de gran valor para la biodiversidad. Evaluamos la efectividad de las TPI en la reducción de la pérdida de la biodiversidad dentro y fuera de las AP mediante la cuantificación de la pérdida de la cobertura arbórea entre el 2000 y 2019 en las ACB a niveles nacional e internacional. También comparamos esta efectividad con las pérdidas en sitios equivalentes fuera de las TPI mapeadas. Con base en una muestra emparejada de celdas de 1-km$^2$ en ACB dentro y fuera de las TPI mapeadas, la pérdida de la cobertura arbórea en las ACB ubicadas fuera de las AP fue menor dentro de las TPI que fuera de ellas. Al contrario, la pérdida en las ACB ubicadas dentro de las AP fue menor afuera de las TPI que adentro de ellas (aunque la diferencia fue por mucho menor). Las tasas nacionales de pérdida de la cobertura arbórea en las ACB variaron sobremanera en relación con su estado en las TPI y en las AP. En la mitad de los 44 países que analizamos individualmente no hubo una diferencia significativa en la tasa de pérdida de la cobertura arbórea en las ACB dentro y fuera de las TPI mapeadas. Las razones detrás de esta variación entre los países podrían aclarar la importancia que tienen las TPI para cumplir con la meta del Convenio sobre Diversidad Biológica de conservar el 30% del suelo para el 2030. La acción coordenada de los gobiernos será crítica para fortalecer y hacer cumplir los derechos de los pueblos indígenas, asegurar su sistema colectivo de tenencia y gobierno, y reconocer sus objetivos para sus tierras y el futuro.

**PALABRAS CLAVE**

áreas protegidas, conservación in situ, deforestación, pérdida de bosque, pueblos indígenas

**【摘要】**

原住民土地 (Indigenous Peoples' lands, IPL) 至少占地球陆地面积的3800万平方公里 (28.1%) 。这些土地对于生物多样性保护至关重要。约 20.7% 的 IPL 分布在受政府保护的地区 (保护地) 。 IPL 中许多对生物多样性具有重要意义的地点可以为全球基于地点的保护目标做出巨大贡献, 但迄今为止尚未得到量化。关键生物多样性区域 (key biodiversity areas, KBAs) 是全球最大的系统性确定对生物多样性具有重要意义的地点的网络。本研究通过量化2000年至2019年间全球和国家级 KBA 的树木覆盖丧失, 并将其与 IPL 外的类似位点进行比较, 评估了 IPL 在减缓保护区内外生物多样性丧失方面的有效性。基于 IPL 内外 KBA 中1平方公里单元的匹配样本, 我们发现保护地外 KBA 的树木覆盖丧失率在 IPL 内低于 IPL 外。相比之下, 保护地内 KBA 的树木覆盖丧失率则在 IPL 外低于 IPL 内 (尽管差异要小得多) 。各国 KBA 的树木覆盖丧失率有很大差异, 与 IPL 和保护地的等级有关。在我们独立分析的 44 个国家中, 有一半国家 IPL 内外 KBA 的树木覆盖丧失率没有显著差异。这种国家间差异的原因可以证明 IPL 在实现《生物多样性公约》提出的"到 2030 年保护 30% 土地"目标中的重要性。为此, 各国政府必须采取协调一致的行动, 加强和落实原住民的权利, 保障他们的集体使用权和治理制度, 并承认他们对其土地和未来的期望。**【翻译: 胡怡思; 审校: 聂永刚】**

**关键词:** 森林砍伐, 森林丧失, 原住民, 保护地, 基于地点的保护

## INTRODUCTION

Site-based conservation is a cornerstone of global biodiversity conservation (CBD, 2022). The conservation of as many as 20% of birds, mammals, and amphibians largely depends on single sites, 62% depends on multiple sites, and 18% on both sites and landscape or seascape scale efforts (Boyd et al., 2008). Key Biodiversity Areas (KBAs) represent the largest global network of sites of significance for the global persistence of biodiversity, identified nationally using standardized quantitative criteria related to threatened or geographically restricted species and ecosystems, ecological integrity, or irreplaceability (IUCN, 2016). Over 16,000 KBAs have been recognized to date (www.keybiodiversityareas.org).

Many KBAs are formally conserved in protected areas (PAs), areas set aside by governments for conservation; 19.6% of KBAs are completely within PAs and 61.0% are completely or partially within them (Key Biodiversity Areas, 2022). However, PAs tend to be in remote and inaccessible locations (Joppa & Pfaff, 2011); often fail to include the most, or most threatened, biodiversity (Beresford et al., 2013; Venter et al., 2014); and are often ineffectively managed (Geldmann et al., 2013) or even inequitable to people (Fletcher et al., 2021). Designation of sites as PAs has sometimes been controversial where this designation affects Indigenous Peoples. In some cases, PA establishment has resulted in, for example, the eviction and displacement of Indigenous communities, loss of traditional management practices, criminalization and restrictions of livelihood activities and access to culturally valued resources, and uncompensated loss of livelihoods among other impacts (Colchester, 2004; Oldekop et al., 2016; Tauli-Corpuz et al., 2020; Whyte, 2018). Such pressures have often generated intergenerational trauma and reduced cultural engagement, leading to declines in peoples' well-being (Fernández-Llamazares, Lepofsky, et al., 2021; Lyver et al., 2019; Zahran et al., 2015). Other effective area-based conservation measures (OECMs) are recognized as an alternative approach to site-based conservation, but concerns have been raised about whether sites proposed or formally recognized as OECMs will effectively conserve biodiversity or whether this recognition is culturally appropriate in Indigenous Peoples' contexts (ICCA Consortium, 2022). For example, rates of tree cover loss are much lower inside PAs than outside, with losses inside potential OECMs similar to that in equivalent matched sites (Donald et al., 2019). Nevertheless, OECMs capture a diverse range of measures that can potentially benefit biodiversity (IUCN-WCPA Task Force on OECMs, 2019). For example, Luther et al. (2021) found that, in a sample of countries, 30% of unprotected or partially protected Alliance for Zero Extinction (AZE) sites (sites that hold the entire population of a critically endangered or endangered species [AZE, 2023]) may potentially qualify as OECMs.

Indigenous Peoples manage or have legal rights to many sites that qualify, or could qualify, as OECMs, as they do for many sites formally designated as PAs. However, many Indigenous Peoples' organizations advocate for the recognition of Indigenous and traditional territories in their own right, as an alternative to OECMs and PAs (Cariño & Farhan Ferrari, 2021). Indigenous Peoples' lands (IPL) identifiable on public maps in 2018 encompassed at least one quarter of the Earth's land (Garnett et al., 2018) and about 36% of the world's intact forest landscapes (Fa et al., 2020). Although the case for the global significance of Indigenous stewardship has been developed by Indigenous scholars and philosophers for decades, if not longer (e.g., Salmón, 2000; Umeek, 2011), scientists have recently started to characterize, quantitatively and qualitatively, biodiversity patterns in IPL (Fernández-Llamazares, López-Baucells, et al., 2021; Schuster et al., 2019). For example, rates of loss of native vegetation (Alves-Pinto et al., 2022) and tree cover loss in IPL appear to be lower than rates outside mapped IPL, although there is considerable spatial variation in patterns (Sze et al., 2021, 2022). The conservation values of IPL have been documented for individual countries (Australia [Renwick et al., 2017]), regions (Amazon [Walker et al., 2020), biomes (tropical dry forests [Pratzer et al., 2023]), and specific taxonomic assemblages (mammals [O'Bryan et al., 2021]; primates [Estrada et al., 2022]). Such lands could therefore play a critical role in the conservation of sites of global biodiversity importance, but more needs to be known if they are to be fully considered in policy forums.

The extent to which IPL and sites of particular importance for biodiversity overlap is unknown. It is also unclear how trends in environmental quality vary among sites on IPL, among sites inside PAs, and where the IPL and PA governance or management is combined. Understanding the influence of different governance and management systems on sites of particular importance for biodiversity (such as KBAs) is crucial for designing approaches to achieve the aims of the Kunming–Montreal Global Biodiversity Framework, in which target 3 commits parties to "[e]nsure…that by 2030 at least 30 per cent of terrestrial, inland water, and of coastal and marine areas…are effectively conserved and managed through ecologically representative, well-connected and equitably governed systems of PAs and OECMs, recognizing indigenous and traditional territories…and integrated into wider landscapes, seascapes and the ocean, while ensuring …sustainable use…is fully consistent with conservation outcomes, recognizing and respecting the rights of Indigenous Peoples and local communities, including over their traditional territories" (CBD, 2022) (building on target 11 of the Aichi targets [CBD, 2010]). Because KBAs are the largest, systematically identified network of sites of particular importance for biodiversity, their effective conservation is key to meeting target 3. Development of an indicator showing the degree to which KBAs are covered by protected and conserved areas is recommended in the Global Biodiversity Framework for target 3 (CBD, 2022).

We conducted spatial analysis of KBAs and IPL for countries for which IPL have been mapped. Primarily, we tested the null hypothesis that the rate of tree cover loss in forest KBAs in IPL is no different from the rate in similar areas outside known IPL. We focused on tree cover because its extent can be mapped with reasonable accuracy, and related change data are available globally (Hansen et al., 2013). However, KBAs contain all habitat types, not just forests. We used matching and generalized linear modeling to attempt to control for potentially confounding effects. Matching is used widely in assessing the impact of interventions, especially at the site scale (e.g., Ribas et al., 2021). We used matching so that we could produce sets of data on tree cover loss inside IPL and in areas outside known IPL that were more similar in terms of characteristics that could influence propensity for tree cover loss than would be the case had we considered all tree cover data. These data were then used in a generalized linear model to test whether there were differences in tree cover loss related to the interaction between IPL and PA status.

# METHODS

## Overlap between KBAs and IPL

We used data on the spatial boundaries of all terrestrial KBAs by selecting KBAs with their system coded as terrestrial in the attribute table (BirdLife International, 2020). The KBA boundary data were converted to a cylindrical equal area projection and spatially intersected with IPL mapped by Garnett et al. (2018) to quantify the area of overlap of each KBA with IPL. Overlaps of <2% of any KBA were assigned a value of zero to account for spatial mapping uncertainty. Overlaps of ≥2% were summed to estimate the total area and proportion of the KBA network that intersected with IPL. We mapped the KBA overlap with IPL with ArcGIS (ESRI, 2020). Data were summarized at the geographic regional level following the United Nations classifications of state boundaries (UN Secretariat, 2022) and at the country level with ISO3 codes.

## Quantification of tree cover loss in KBAs in relation to IPL and PAs

Quantification of tree cover loss was undertaken in forest KBAs. Forest KBAs were identified by filtering KBAs with >2% overlap with mapped IPL, selecting those that had been identified for at least 1 forest-dependent qualifying species (i.e., species that qualify the site as a KBA) (Key Biodiversity Areas Partnership, 2020). Forest-dependent species were defined as species for which forest is the only type of habitat listed for them on the IUCN Red List (BirdLife International, 2019; IUCN, 2019). We assessed PA boundaries with data from Protected Planet (WDPA) (UNEP-WCMC, 2020) filtered to exclude PAs with a status of proposed and not reported and UNESCO's biosphere reserves; otherwise, all PA categories I–VI were retained (following UNEP-WCMC [UNSD, 2022]).

Tree cover loss was quantified at the level of a 1-km$^2$ grid produced in ArcGIS (ESRI, 2020). The 1-km$^2$ grid was overlaid on the KBA, IPL, and PA layers. Cells that partially overlapped a KBA, IPL, or PA boundary were discarded. Areas falling outside the IPL layer were considered outside mapped IPL to reflect the fact that they may contain IPL that were not mapped or recognized as such at the time of publication of Garnett et al. (2018).

Data on tree cover in 2000 and tree cover loss from 2001 to 2019 were extracted for each of the retained 1-km$^2$ cells with Global Forest Change 1.7 data, originally described by Hansen et al. (2013), accessed via Google Earth Engine (Gorelick et al., 2017). The native spatial resolution of the data of Hansen et al. (2013) is 30 m. For each of the 1-km$^2$ cells, we calculated the number of 30-m tree cover pixels that had ≥50% canopy cover in 2000 to estimate the total tree cover in 2000 (some therefore also included nonforested areas). We then estimated the number of tree cover pixels lost from 2001 to 2019 by calculating the number of pixels lost per cell each year (because the data are structured with a layer of pixels lost for each year) and then summed these to generate the total number of pixels lost over the 19-year period. Cells with no tree cover in 2000 were excluded.

## Matching

Matching has become an established tool in conservation studies for estimating counterfactuals (Ribas et al., 2021; Schleicher et al., 2020). It can be used to identify controls for comparison with treatments so that both sets of data are similar in terms of potentially confounding effects. We used matching to generate a similar set of cells inside and outside mapped IPL with propensity score matching and the nearest neighbor method (producing 1-to-1 matched pairs). Analyses were conducted at 2 levels. First, we undertook a global analysis of the combined KBA data across the countries with sufficient data after overlapping the 1-km$^2$ grid with KBAs, IPL, and PAs. Second, we undertook national-level analyses for each country in turn. For both the global- and national-level analyses, matching was used to reduce the potential for systematic differences in the locations of KBAs and IPL to affect the results. Matching areas elsewhere that had similar values for variables describing the geography, vegetation levels and type, country, and governance allowed us to distinguish the effect of IPL on loss of tree cover in forest KBAs (Ribas et al., 2021; Schleicher et al., 2020).

All cells at which there was >50% tree cover in countries for which IPL data were available (Garnett et al., 2018) were considered in matching. The matching was carried out for the entire data set for the global analysis and repeated for each country for the national-level analysis. Before matching, there were 2,176,960 1-km$^2$ grid cells across 64 countries (791,969 in KBAs inside IPL, 1,384,991 in KBAs outside mapped IPL). After matching, there were 668,906 cells across 50 countries (334,453 cells inside IPL and outside mapped IPL). The variables we used in our matching were extracted at the 1-km$^2$ level. Global variation in the type of forest in each cell was determined by overlapping with GLC2000 (Bartholome & Belward, 2005), a data set contemporary with the baseline year of 2000, which classifies forest into 10 types (Table 1). We used Google Earth Engine to extract data on the mean slope in degrees, mean elevation in meters (U.S. Geological Survey, 2010), mean accessibility (Weiss et al., 2018), and proportional tree cover within a 5-km radius in 2000 (with the tree cover in year 2000 layer from version 1.7 of the Global Forest Change data) (Gorelick et al., 2017).

For matching, we used the package MatchIt (Ho et al., 2011) in R (R Core team, 2021). Cells in KBAs inside IPL were matched to cells in KBAs outside mapped IPL. We matched with the caliper set to 0.5 SD to increase the similarity in the matching variables between data sets inside and outside mapped IPL. We used the categorical variables country, PA status (inside or outside a PA), forest type and the continuous variables tree cover in 2000, mean slope, mean elevation, mean accessibility (time to access a location; the higher the number, the lower the accessibility), and proportional tree cover within a 5-km radius in 2000. We matched exactly for categorical variables and matched based on minimum distance for the continuous variables. We ran 1 match in all cases.

## Modeling

A negative binomial model (run in R package "lme4" [Bates et al., 2015]) was used to assess whether tree cover loss differed between 1-km$^2$ cells in KBAs inside IPL and those in KBAs outside mapped IPL. Whether each 1-km$^2$ cell was inside or outside a PA was included in the model; this enabled us to consider the effect of PAs on tree cover loss. The model treatments were inside mapped IPL and outside PA; outside mapped IPL and inside PA; and inside both mapped IPL and PA. All were compared against a control of outside both mapped IPL and PA.

**TABLE 1** Summary of negative binomial mixed-effect model of tree cover loss from 2001 to 2019 in 1-km$^2$ cells in Key Biodiversity Areas across all countries where Indigenous peoples' lands (IPL) were identified by Garnett et al. (2018).

| Variable | Estimate (SE) | *Z* (*p*) |
|---|---|---|
| Intercept | −3.92 (0.06) | −70.70 (<0.001) |
| Inside IPL | −0.17 (0.02) | −9.89 (<0.001) |
| Inside protected area (PA) | −1.31 (0.02) | −75.34 (<0.001) |
| Inside IPL × inside PA | 0.25 (0.02) | 11.61 (<0.001) |
| Mean tree cover 2000 in 5 km buffer* | −0.62 (0.01) | −48.67 (<0.001) |
| Mean accessibility* | −0.73 (0.01) | −86.48 (<0.001) |
| Mean elevation* | −0.37 (0.01) | −47.46 (<0.001) |
| Mean slope* | −0.09 (0.01) | −13.61 (<0.001) |
| Mean tree cover 2000* | −0.06 (0.01) | −7.72 (<0.001) |
| Shrub cover, closed-open, deciduous | −0.06 (0.06) | −1.02 (>0.050) |
| Shrub cover, closed-open, evergreen | 0.26 (0.05) | 5.56 (<0.001) |
| Tree cover, broadleaved, deciduous, closed | −0.51 (0.05) | −11.04 (<0.001) |
| Tree cover, broadleaved, deciduous, open | −0.55 (0.07) | −7.78 (<0.001) |
| Tree cover, broadleaved, evergreen | −0.14 (0.04) | −4.00 (<0.001) |
| Tree cover, mixed leaf type | −0.3 (0.09) | −3.38 (<0.001) |
| Tree cover, needle-leaved, deciduous | −2.51 (1.37) | −1.83 (>0.050) |
| Tree cover, needle-leaved, evergreen | −0.16 (0.06) | −2.64 (<0.010) |
| Tree cover, regularly flooded, fresh water | −0.52 (0.04) | −11.68 (<0.001) |
| Tree cover, regularly flooded, saline water | −0.26 (0.13) | −1.96 (<0.050) |

*Note*: Continuous variables are marked with an asterisk (*); remaining are factors.

In the global model, the number of tree cover pixels lost per cell was modeled as a function of all of the matching variables (listed above), which were fitted as covariates (tree cover in 2000, mean slope, mean elevation, mean accessibility, proportional tree cover in 5-km radius in 2000), fixed effects (inside or outside mapped IPL, PA status, forest type), or random factors (KBA site identity and country). An interaction term was also included between inside and outside mapped IPL and PAs. The log of the total number of tree cover pixels (in 2000) was included as an offset to weight the tree cover loss by the initial tree cover.

For the national-level analysis, countries were filtered to include only those that had a sample size of at least 30 pixels inside IPL and outside mapped IPL prior to matching and those for which models converged. This resulted in 44 countries in the analyses (Appendix S6). The national-level model included inside and outside each of the IPL and PAs (both as binary fixed effects); the interaction term between inside and outside IPL and PAs; KBA site identity as a random factor; and the offset of the total number of forest pixels. Categorical variables were excluded because sample sizes were small and categorical variables had an exact match from the earlier matching. We calculated the number of countries that had significantly higher or lower levels of tree cover loss in their KBAs or that had no significant difference in the rate of tree cover loss for inside versus outside mapped IPL (both outside PA); inside versus outside PA (both outside mapped IPL); and inside IPL and PA versus outside both. Moran's *I* was used to quantify spatial autocorrelation in the residuals of this model. We used a fixed distance band of 30 km to enable the model to deal with the large number of data points. We repeated the analysis for the global model and the residuals of the global model by country. We undertook analyses in ArcMap 10.5.1 (ESRI, 2022).

Interaction plots were produced using cat plot from the R package interactions (Long et al., 2019) to show the effect that being inside IPL, inside a PA, or inside both had on forest loss in KBAs compared with outside both mapped IPL and PAs. For all countries where KBAs inside IPL showed a significant reduction in tree cover loss, the mean effect size was calculated from the parameter estimates for models.

# RESULTS

## Overlap between KBAs and IPL

In the 83 countries for which IPL were mapped, IPL covered at least 3.73 million km$^2$ across 2705 (33.6%) KBAs (Figure 1), which is equivalent to 35.6% of the KBA in these countries and a mean coverage of 22.4% per KBA (including those with no overlap with IPL). This equates to approximately 28% of the total area of land in all KBAs globally. Around 14% of the total area of land in KBAs globally was in mapped IPL and outside PAs. The distribution of KBAs inside IPL varied regionally and nationally, with the highest percentage of KBA area within mapped IPL and the greatest proportional area of IPL

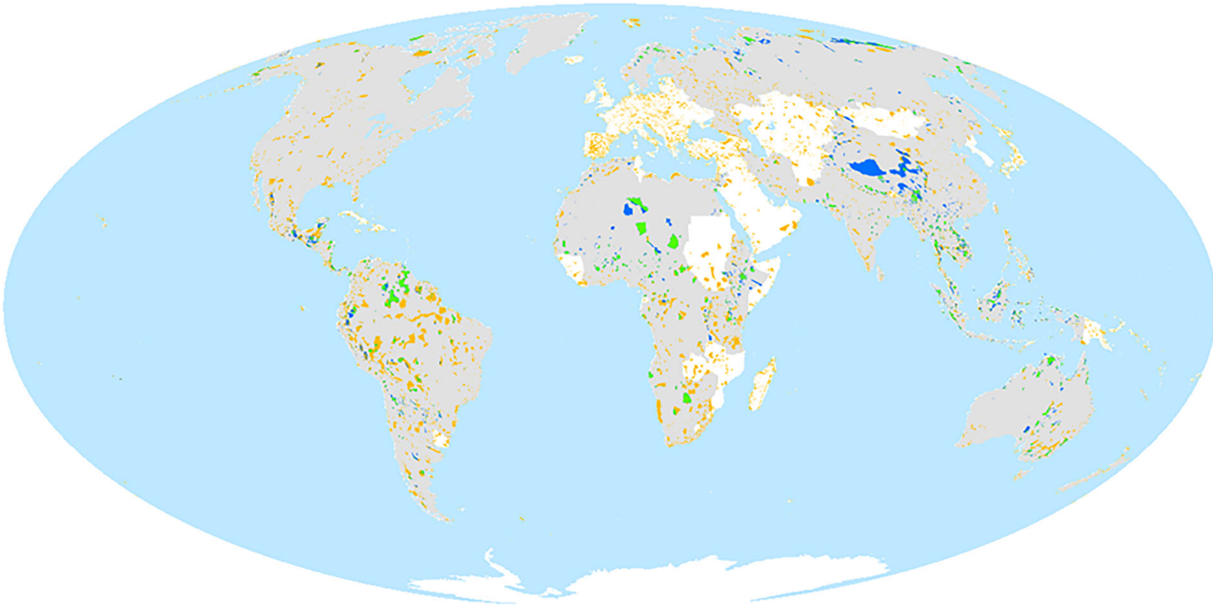

**FIGURE 1** Global distribution of terrestrial Key Biodiversity Areas (KBAs) relative to Indigenous Peoples' lands (IPL) and protected areas (PA) (blue, inside IPL and outside PAs; green, inside IPL and inside PAs; orange, outside mapped IPL; gray, countries where IPL are reported; white, countries where IPL are not reported or where data are unavailable). The KBAs with <2% of their area overlapping an IPL or a PA are treated as not covered by the KBA.

in East and Southeast Asia (Appendices S3 & S4, respectively). Burkina Faso and Mali had the highest coverage of their KBA networks within mapped IPL (100% of 10 and 17 KBAs; 12,962 and 24,470 km$^2$, respectively), and South Africa had the lowest (0.6% of 169 KBAs; 9575 km$^2$). China had the largest area of IPL reported in KBAs (0.78 million km$^2$ across 614 KBAs; 32.3%) (China does not consider its 55 ethnic minorities Indigenous peoples), and Libya had the smallest area (3.68 km$^2$ across 18 KBAs; 5.6%).

## Quantification of tree cover loss in KBAs in relation to IPL and PAs

The global analysis that considered matched data from all (64) countries together indicated that loss of tree cover in KBAs from 2001 to 2019 was lower inside IPL than at matched locations in KBAs that were outside mapped IPL (Table 1). The significant interaction between IPL and PA (Table 1) indicated that IPL and PA status correlated with tree cover loss in KBAs. The interaction plots (Figure 2) based on the output from the generalized linear models (Table 1) showed that tree cover loss was lower in KBAs that were outside PAs and inside IPL than in KBAs outside both PAs and mapped IPL. However, for KBAs in PAs, tree cover loss was lower outside mapped IPL than inside IPL (although the difference was far smaller). There was substantial spatial autocorrelation in the residuals of this global model (Moran's $I = 0.069$, $z = 1086.62$, $p < 0.001$). This was not due to any 1 country because there was significant ($p < 0.05$) spatial autocorrelation in residuals for all but 8 of the 44 countries with at least 30 pixels inside IPL and outside mapped IPL before matching.

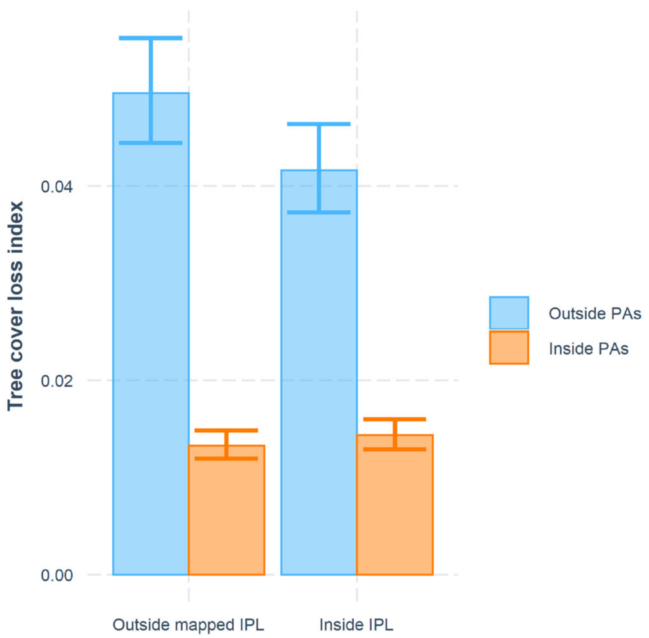


**FIGURE 2** Tree cover loss rate in Key Biodiversity Areas (KBAs) inside and outside mapped Indigenous Peoples' lands (IPL) and inside or outside protected areas (PAs) (error bars, 95% confidence intervals). Results are from a negative binomial model run on matched locations in KBAs in countries where IPL were identified by Garnett et al. (2018) (Table 1).

The national-level analyses indicated there was substantial variation in the patterns of tree cover loss in KBAs in the 44 countries with at least 30 cells inside IPL and outside mapped IPL before matching (Figure 3; Appendices S5 & S6). In comparison with KBAs outside mapped IPL and outside PAs, we found that tree cover loss was significantly lower in KBAs

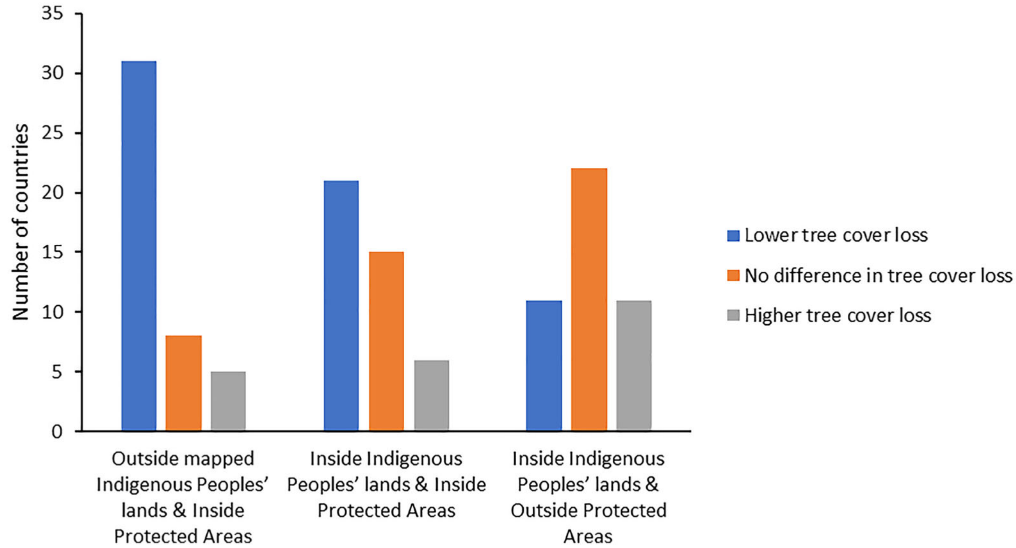


**FIGURE 3** Number of countries that have statistically significant higher or lower (or no significant difference) levels of tree cover loss in their Key Biodiversity Area (KBA) network over all inside Indigenous Peoples' lands (IPL), protected areas (PAs), or both compared with KBAs inside and outside mapped IPL and PAs. Appendices S5 and S6 contain a full breakdown of the results. The total number of countries was 44, except for 2 countries (French Guiana and Morocco) that had no interaction between IPL and PAs, so total number of countries was 42 for inside IPL and PAs.

outside mapped IPL that were inside PAs in most (70%, 31) countries. Tree cover loss was significantly lower in KBAs that were inside IPL and inside PAs in half (21) of the countries compared with KBAs outside mapped IPL and PAs (50%). We detected no significant difference in tree cover loss in KBAs that were inside IPL but outside PAs compared with KBAs that were outside mapped IPL and outside PAs in half (22) of the countries (50%).

See Appendix S1 for overall matching results (with all data) and Appendix S2 for national-level matching results, where the majority of mean values for each covariate inside and outside mapped IPL were more similar following matching.

## DISCUSSION

Our results showed there are large areas in the KBA network that are not in PAs recognized by the state, where Indigenous Peoples have rights recognized by the state or where Indigenous Peoples retain a substantial de facto influence on management. Retaining the biodiversity values of these areas will be necessary if global aspirations to conserve 30% of land, particularly areas of importance for biodiversity, are to be achieved.

Our finding that tree cover loss outside PAs was lower on parts of KBAs inside IPL than outside mapped IPL is consistent with results of other studies showing the important role of Indigenous Peoples in reducing tree cover loss (Pratzer et al., 2023; Sze et al., 2021). The wide variation in patterns of tree cover loss among countries shows that cause and effect need to be determined on a site-by-site basis. However, the overall result of lower tree cover loss in KBAs in IPL suggests that IPL could play a major role in the conservation of sites of biodiversity importance and, as such, could make an important contribution to achieving the Global Biodiversity Framework targets for site-based conservation.

Our results add to growing evidence that a substantial share of the world's biodiversity lies within IPL (Estrada et al., 2022; O'Bryan et al., 2021) and is highly dependent on Indigenous Peoples' stewardship practices, knowledge systems, and cultural connections to their lands (Brondizio et al., 2021). Many studies show the myriad of contextual factors that influence the ability of IPL to mitigate forest loss, such as land tenure security (Baragwanath & Bayi, 2020), Indigenous knowledge maintenance and revitalization (Fernández-Llamazares, Leposky, et al., 2021), and strong Indigenous-led governance (Artelle et al., 2019). The wide variation in rates of tree cover loss in IPL across countries is at least to some extent a reflection of the diversity of Indigenous Peoples' sociocultural realities vis à vis rapidly expanding deforestation frontiers (Buchadas et al., 2023; Carneiro da Cunha & de Almeida, 2000). Although Indigenous communities are proactively combatting forest loss in many IPL through their millennia-old stewardship systems and cultural practices (e.g., Mistry et al., 2016), in other contexts IPL are increasingly vulnerable to illegal deforestation (e.g., Silva-Junior et al., 2023). Moreover, there is well-documented evidence that IPL are an increasing target for extractive and industrial development (Owen et al., 2023; Scheidel et al., 2023). The rapid expansion of extractive and commodity frontiers into IPL could jeopardize the conservation value of such lands and exacerbate legacies of land dispossession (Farrell et al., 2021; Kennedy et al., 2023). Coordinated action to support Indigenous Peoples in safeguarding their lands, and recognizing their historical rights to do so, is therefore inextricably linked to global efforts to address biodiversity loss (IPBES, 2019).

About one half of the land inside KBAs and mapped IPL was inside PAs. Our global analyses indicated that tree cover loss in

KBAs was lowest in PAs, regardless of whether they were inside or outside mapped IPL. Although there are multiple reasons for this pattern, perhaps the most obvious is that many PAs have some level of governmental backing to protect conservation values, whereas states often sanction multiple land uses of IPL (although some PAs do allow more extractive activities than may be typical in IPL). However, the benefit of PAs is not universal, probably because of national-level variation in the effectiveness of protection (e.g., Leverington et al., 2010; Rife et al., 2013). Of the countries with at least 30 pixels inside IPL and outside mapped IPL before matching, 5 had higher deforestation rates inside than outside PAs, and in 8 countries there was no effect. The capacity of conservation agencies to resist pressures, such as from those causing deforestation, and the effectiveness of conservation interventions are often compromised by politics and corruption (Schleicher et al., 2019; West et al., 2006). This may also reflect different PA concepts among countries (e.g., PAs in the United Kingdom are very different from those in Brazil), so PA effectiveness varies among (and within) countries too (Cazzolla Gatti et al., 2023; Geldmann et al., 2014).

Notwithstanding deficiencies in PA management, conserving forest on IPL outside PAs is potentially much more difficult. A lack of state protection can mean that KBAs on IPL outside PAs can more readily be targeted by people destroying forests than those in areas formally allocated to conservation (e.g., dos Santos et al., 2022; Siqueira-Gay & Sánchez, 2021; Siqueira-Gay et al., 2020; Urzedo & Chatterjee, 2021). In such cases, Indigenous Peoples are often at the forefront of resistance and the last line of defense against environmental degradation (Armstrong & Brown, 2019; Spice, 2018), despite their being subject to higher levels of arrest, violence, and even death than other environmental defenders (Beattie et al., 2023; Scheidel et al., 2020). Also, unlike in most PAs, many IPL contain patches of forest that have been formerly cleared as part of traditional swidden agriculture to grow food crops (Ziegler et al., 2011). Such forests may recover their biodiversity values during extended fallow periods, but our methods were not designed to detect forest recovery. Finally, some Indigenous Peoples, or their leaders, have specific socioeconomic aspirations and may actively wish to clear forests for profit, whether because they are actively espousing such conversion themselves (Hicks et al., 2015) or through the influence of external actors operating on their lands (Foster, 2015; He et al., 2019).

Although the balance of these reasons will vary by case, the lack of recognition, respect, and enforcement of Indigenous Peoples' rights is usually the most powerful driver of deforestation (Baragwanath & Bayi, 2020). This could be one reason behind the pattern we found in the national-level analysis, where the majority of countries showed no effect of IPL on tree cover loss in KBAs, whereas the majority of countries experienced lower rates of tree cover loss in KBAs when inside (compared with outside) PAs (whether inside or outside mapped IPL). From the advent of colonialism to the present, Indigenous Peoples have a history of disempowerment in states that have asserted ownership of their traditional lands. Many Indigenous Peoples often have no legal ownership or tenure rights over their traditional lands and, for many of the countries where losses of forest have been greatest on IPL outside PAs, disenfranchisement has been strongly manifest in recent decades (e.g., Cameroon, Beckline et al., 2022; Bryan, 2019; Nicaragua, Betts et al., 2020). Even where IPL are managed by Indigenous Peoples with a conservation aim, governance and contextual factors, such as incentives for mining and access and use of IPL, might also lead to deforestation. That there are any countries in which the rate of forest loss from KBAs on IPL is lower than on other lands is notable given the frequency with which Indigenous Peoples' rights are ignored and violated (International Work Group on Indigenous Affairs, 2022).

Given the pressures on forests, the extent to which IPL will retain forests in future depends on multiple factors. These include the extent to which Indigenous Peoples' aspirations for the lands they control accord with conservation aims; extent to which biodiversity values are retained as a direct consequence of Indigenous Peoples' stewardship and governance of those lands; and, in locations where the influence of Indigenous Peoples' management has been eroded, the extent to which governments are willing to return rights and to control causes of forest loss that are not sanctioned by Indigenous Peoples.

With respect to the first of these points, the interests of Indigenous Peoples may not match exactly those who base their conservation advocacy primarily on current Western science (Lyver et al., 2014). Although retention of forest to benefit biodiversity can be an emergent property of Indigenous Peoples' cultural and spiritual approaches to land stewardship, active management for the purpose of conservation may not always be the underlying cause.

Second, IPL often overlap other tenures making it difficult for Indigenous Peoples to exercise control over management. In many jurisdictions, Indigenous Peoples are not fully empowered politically and economically to express their perspectives in an equitable manner (International Work Group on Indigenous Affairs, 2022). For example, although the post-2020 Global Biodiversity Framework's aspirations include "recognising the rights of Indigenous Peoples and local communities, including over their traditional territories…" (CBD, 2022), the rights of Indigenous Peoples are formally recognized in <1% of terrestrial PAs even though 40% overlap IPL (Reyes-García et al., 2022).

Third, in countries where Indigenous Peoples' rights are violated, the conservation of KBAs in IPL faces additional challenges. There is an ongoing history of conservationists working with state authorities to protect biodiversity without recognition of Indigenous Peoples, often undermining long-term sustainability of such conservation efforts (Fletcher et al., 2021). The long-term impacts of colonial policies on conservation generally result in the disenfranchisement, marginalization, and exclusion of Indigenous communities, creating conflicts that ultimately undermine the ecological condition of such sites (Domínguez & Luoma, 2020). A major challenge to conservation of KBAs and conservation more generally on IPL is to avoid such outcomes (Brockhaus et al., 2021).

Given these factors, it is not straightforward to determine whether KBAs on IPL that are currently outside PAs would be best conserved through recognizing OECMs, designating new (or expanding existing) PAs, or recognizing the integrity and distinct nature of IPL beyond categories of PA and OECMs (as often called for by Indigenous Peoples' organizations [Cariño & Farhan Ferrari, 2021]). Such determinations will be highly context specific. Whatever the governance system, it is critical that areas should only count toward achievement of target 3 if they demonstrate effective biodiversity outcomes. This is explicit in the name and definition of OECMs, but many PAs are currently ineffective.

## Limitations

Our analysis was restricted to forest KBAs and so did not consider the impact of IPL on KBAs containing other habitats. We also used satellite imagery of tree cover loss derived by Hansen et al. (2013) to consider conservation impact, defined as a reduction in the rates of tree cover loss. Although forest retention is important to the forest species' populations for which these KBAs were identified, tree cover data may miss more nuanced changes below the canopy and hence fail to capture, for example, degradation of forest condition or unsustainable hunting. We did not consider historic tree cover loss and the previous influence of IPL because data were unavailable before 2000. The tree cover data set also does not mask out plantations—some areas of apparent tree cover loss may result from felling of plantations or may mask replacement of forest by plantations, each with opposing implications for forest-dependent species. We also did not consider forest regrowth or restoration because of the lack of an equivalent reliable tree cover gain layer over the same period that excluded plantations. However, such a short period of regenerative growth is unlikely to be sufficient to create habitat for forest-dependent species almost all of which require older forests.

The statistical matching process led to exclusion of many areas in KBAs, so many grid cells were excluded because they could not be statistically matched to equivalent sites outside the IPL boundaries or they did not have a large enough sample size after matching. This also meant that some countries had lower sample sizes, meaning many countries analyzed at the global scale could not be analyzed at a national scale. After the matching and the modeling, there was considerable spatial autocorrelation in the residuals of the model. We did attempt to control for this by the inclusion of KBA identity as a random effect in the model to control for nonindependence, but substantial autocorrelations remained in the residuals. This residual nonindependence of proximal data points means that the results need to be treated with some caution because spatial autocorrelation can potentially affect the estimates of site conservation effectiveness (Negret et al., 2020).

The IPL data set was incomplete, and there was uncertainty in the exact boundaries of the IPL. Further details on the limitations of this data set are in Garnett et al. (2018). We acknowledge that blank areas in the IPL data set do not necessarily reflect a lack of Indigenous Peoples or their lands; rather, they represent areas for which ownership or management by Indigenous Peoples cannot be inferred based on the publicly accessible geospatial data compiled by Garnett et al. (2018). The KBA and PA data sets were also incomplete, given that KBAs have not been identified for all taxa, ecosystems, and other biodiversity features in all locations and that not all PA boundaries are publicly available or mapped as polygons. Boundaries for KBAs and PAs may include spatial errors, although marginal boundary issues should not have confounded our results because we only included areas that fell wholly inside or outside KBAs, IPL, and PAs.

## Further work

It is critically important that further work explore why tree cover loss (particularly in KBAs) was lower in some IPL than others so as to determine whether this is a result of differences in Indigenous Peoples' rights (including land tenure security), recognition by governments, corruption, past history of tree cover loss elsewhere, changes in traditional stewardship practices, land grabs and encroachment, or direct deforestation by Indigenous Peoples (and if so why). This also needs to be explored in nonforest systems to understand wider impacts on other habitats and species. This will help identify how to enhance the contribution that IPL could make to meeting the ambition of effectively conserving 30% of land, particularly areas of importance for biodiversity, by 2030 while ensuring that Indigenous Peoples' rights and land tenure are fully recognized and enforced.

We focused on land-cover change in IPL. Leakage, whereby negative activities, such as tree cover loss, that would otherwise have occurred in a site are displaced to another location that is unmanaged or protected, can present a problem when estimating site-based conservation effectiveness (Ewers & Rodrigues, 2008) and may ultimately undermine the impact of site-based conservation (Ford et al., 2020). However, displacement of deforestation is less likely to be a problem for IPL given that connections to the wider economy from which leakage would occur tend to be weaker (Pratzer et al., 2023). Quantification of leakage from IPL is an important next step to determine the net impact of IPL on tree cover and hence biodiversity conservation.

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## SUPPORTING INFORMATION

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

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