Rates of tree cover loss in key biodiversity areas within 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 the global 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, both within and outside PAs, by quantifying tree cover loss 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 of mapped IPL, tree cover loss in KBAs outside of PAs was lower inside IPLs when compared to outside mapped IPLs. By contrast, tree cover loss in KBAs inside of PAs was lower outside mapped IPLs than inside IPLs (although the difference was far smaller). However, national rates of tree cover loss in KBAs varied greatly in relation to their IPL and PA status. In 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 help understand the importance of IPL in meeting the CBD'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.

Introduction

Site-based conservation is a cornerstone of global biodiversity conservation (CBD, 2022a). The conservation of as many as 20% of birds, mammals and amphibians largely depend on single sites, 62% depend on multiple sites and 18% on both sites and landscape/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 standardised quantitative criteria relating 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 through Protected Areas (PAs), areas set aside by governments for conservation, with 19.6% of KBAs completely within PAs and 61.0% completely or partially within them; Key Biodiversity Areas, 2022). However, previous studies have found that PAs tend to be in remote and inaccessible locations (Joppa & Pfaff, 2011), that they often fail to include the most, or most threatened, biodiversity (Beresford et al., 2011; Venter et al., 2014) and that their management is often ineffective (Geldmann et al., 2020) or even inequitable (Fletcher et al., 2021). Designation of sites as PAs has sometimes been controversial where it affects Indigenous Peoples. In some cases it has resulted in 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 devastation of livelihoods, among others (Colchester, 2004; Oldekop et al., 2015; Whyte, 2018; Tauli-Corpuz et al., 2020). Such pressures have often generated legacies of intergenerational trauma and reduced cultural engagement, leading to declines in peoples' well-being (Zahran et al., 2015; Lyver et al., 2017; Fernández-Llamazares et al., 2021). Other effective area-based conservation measures (OECMs) are recognised as an alternative approach for site-based conservation, but concerns have been raised about whether sites proposed or formally recognised as OECMs will be genuinely effective for conserving biodiversity, or whether this recognition is culturally appropriate in Indigenous Peoples' contexts (ICCA Consortium, 2022). For example, rates of tree cover loss were found to be much lower inside PAs than outside, with losses inside potential OECMs similar to equivalent matched sites (Donald et al., 2018). 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 with many sites formally designated as PAs. However, many Indigenous Peoples' organizations have advocated for the recognition of Indigenous and traditional territories in their own right, as a third pathway beyond OECMs and PA categories (Cariño and Farhan Ferrari, 2021). Indigenous Peoples' lands (IPL) identifiable in 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). While 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), scientific research has recently started to characterize, both quantitatively and qualitatively, biodiversity patterns in IPL (Schuster et al., 2019; Fernández-Llamazares et al., 2021). For example, rates of loss of "native vegetation" (Alves-Pinto et al., 2022) and tree cover loss within IPL have been reported 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). IPL 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.

One gap in knowledge is the extent of overlap between IPL and sites of particular importance for biodiversity or how trends in environmental quality vary between sites on IPL, those inside PA, and where the IPL and PA governance/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 "Ensure and enable that by 2030 at least 30 per cent of terrestrial, inland water, and of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem functions and services, are effectively conserved and managed through ecologically representative, well-connected and equitably governed systems of PAs and other effective area-based conservation measures (OECMs), recognizing indigenous and traditional territories, where applicable, and integrated into wider landscapes, seascapes and the ocean, while ensuring that any sustainable use, where appropriate in such areas, is fully consistent with conservation outcomes, recognizing and respecting the rights of indigenous peoples and local communities, including over their traditional territories" (CBD, 2022a; building on Target 11 of the Aichi Targets (CBD, 2010)). As KBAs are the largest, systematically identified network of sites of particular importance for biodiversity, they are key priorities to effectively conserve and manage in order to meet Target 3, and 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, 2022a).

Here we describe spatial analysis of KBAs and IPL for countries for which IPL have been mapped. Primarily, we test the null hypothesis that the rate of tree cover loss in forest KBAs within IPL is no different to the rate in similar areas outside mapped IPL. We focus on tree cover as its extent can be mapped with reasonable accuracy, and change data are available globally (Hansen et al., 2013). However, KBAs cover all habitat types, and are not restricted to forests. We use matching and then generalised linear modelling 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). Matching meant that we were able to produce sets of data on tree cover loss from inside IPL and in areas outside mapped IPL that had a greater balance (i.e., were more similar) in terms of a set of characteristics that could influence propensity to tree cover loss than would be the case had we considered all data for which there were tree cover data. These data were then used in a generalised linear model to test whether there have been 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 mapped Indigenous Peoples' Lands (IPL; from Garnett et al., 2018) to quantify the area of overlap of each KBA by mapped IPL. Overlaps of <2% of any KBA were taken to be zero, to account for spatial mapping uncertainty. Overlaps of ≥2% were then summed to estimate the total area and proportion of the KBA network that intersects with IPL, and the KBA overlap by IPL was mapped using ArcGIS (ESRI, 2020). Data were also summarised at the geographic regional level following the United Nations classifications of state boundaries (UN Secretariat, 2022), and at the country level using ISO3 codes.

Quantification of tree cover loss within 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 one 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 habitat listed as suitable in the IUCN Red List (IUCN, 2019; BirdLife International, 2019). PA boundaries were assessed from Protected Planet (WDPA; UNEP-WCMC, 2020), filtered to exclude PAs with a status of 'proposed' and 'not reported', and UNESCO's biosphere reserves, but 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 are referred to as being "outside mapped IPL" to reflect the fact that they may contain Indigenous Peoples' lands that were not mapped or recognised as such at the time of publication of Garnett et al. (2018).

Data on tree cover in 2000 and tree cover loss between 2001 and 2019 were extracted for each of the retained 1-km^2 cells, using version 1.7 of the Global Forest Change 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. Within 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 (note some will therefore also include non-forested 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 (as the data is 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.

Data analysis - matching

Matching has become an established tool in conservation studies for estimating counterfactuals (Schleicher et al., 2020; Ribas et al., 2021). It can be used to identify controls for comparison to treatments so that both sets of data are similar in terms of potentially confounding effects. Here matching was used to generate a balanced set of cells inside and outside mapped IPL, using propensity score matching and the nearest neighbour method (producing one to one matched pairs). Analyses were conducted at two levels. First, we undertook a global analysis on 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 to areas elsewhere that had similar values for variables describing the geography, vegetation levels and type, country and governance allowed us to distinguish the influence of IPL on tree cover loss in forest KBAs (Schleicher et al., 2020; Ribas et al., 2021).

All cells at which there was >50% tree cover within 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 then repeated for each country in turn 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) which reduced to 668,906 cells across 50 countries after matching (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 et al., 2005), a dataset contemporary with the baseline year of 2000, which classifies forest into ten types (Table 1). We extracted data on the mean slope in degrees and mean elevation in metres (GMTED2010, 2010), mean accessibility (Weiss et al., 2018), and proportional tree cover within a 5-km radius in 2000 (using the tree cover in year

2000 layer from version 1.7 of the Global Forest Change data), using Google Earth Engine (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 caliper set to 0.5 standard deviations to increase the similarity in the matching variables between datasets inside and outside mapped IPL. We used the following variables that were either categorical or continuous. The categorical variables were country, PA status (inside or outside a PA), forest type. The continuous variables were tree cover in 2000, mean slope, mean elevation, mean accessibility (measured as time to access a location, so higher numbers indicate lower accessibility) and proportional tree cover within a 5-km radius in 2000. We matched exactly on categorical variables and minimum distance for the continuous variables. We ran one match in all cases. See Appendix S1 for overall matching results (using all data) and Appendix S2 for national level matching results, where the majority of mean values for each covariate inside and outside mapped IPLs showed improvement in balance following matching.

Data analysis - modelling

A negative binomial model (using R package "lme4"; Bates et al., 2015) was used to assess whether tree cover loss differed between 1-km^2 cells within KBAs inside IPL and those within KBAs outside mapped IPL. In addition, whether each 1-km^2 cell was inside or outside a PA was included in the model; this enabled us also to consider the influence of PAs on tree cover loss. The model treatments were (i) inside mapped IPL and outside PA, (ii) outside mapped IPL and inside PA, and (iii) inside both mapped IPL and PA, where all are compared against a control of outside both mapped IPL and PA.

In the global model, the number of tree cover pixels lost per cell was modelled 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/outside mapped IPL, PA status, forest type), or random factors (KBA site ID and country). An interaction term was also included between inside/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 those that had a sample size of at least 30 pixels both inside IPL and outside mapped IPL prior to matching, and where models converged. This resulted in 44 countries being used for analysis (see Appendix S6). The national level model included only inside/outside each of IPL and PAs (both as binary fixed effects), the interaction term between inside/outside IPL and PAs, KBA site ID as a random factor and the offset of the total number of forest pixels. Categorical variables were excluded as we often had small sample sizes and these had an exact match from the earlier matching. We calculated the number of countries that have 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 each of: (i) inside vs outside mapped IPL (both outside PA), (ii) inside vs outside PA (both outside mapped IPL) and (iii) inside IPL and PA vs outside both. Moran's I was used to quantify spatial autocorrelation in the residuals of this model. Analysis 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 analysis 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, within 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

Within the 83 countries for which IPL were mapped, IPL covered at least 3.73 million km^2 across 2,705 (33.6%) KBAs (Figure 1), which is equivalent to 35.6% of the KBA area within these countries and a mean coverage of 22.4% per KBA (including those with no overlap with IPL). This equates to c. 28% of the total area of land within all KBAs globally. Around 14% of the total area of land within KBAs globally is within mapped IPL and outside of PAs. The distribution of KBAs inside IPL varied regionally and nationally, with the highest percentage of KBA area falling within mapped IPL and the greatest proportional area of IPL in East Asia and South-east Asia (Appendix S3 and S4 respectively). Burkina Faso and Mali have the highest coverage of their KBA network within mapped IPL (100% of 10 and 17 KBAs; 12,962 km^2 and 24,470 km^2 respectively), and South Africa has the lowest (0.6% of 169 KBAs; 9,575 km^2). China has the largest area of IPL reported in KBAs (0.78 million km^2 across 614 KBAs; 32.3%, noting that China does not consider its 55 ethnic minorities as Indigenous Peoples) and Libya has the smallest area (3.68 km^2 across 18 KBAs; 5.6%).

Quantification of tree cover loss within 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 between 2001 and 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) indicates that both IPL and PA status correlate with tree cover loss in KBAs. The interaction plots (Figure 2) based on the output from the GLMs in Table 1 show that tree cover loss was lower in KBAs outside PAs inside IPL than in KBAs outside both PAs and mapped IPL. However, for KBAs within 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 one country as there was significant ($P<0.05$) spatial autocorrelation in residuals for all but eight of the 44 countries with at least 30 pixels both inside IPL and outside mapped IPL before matching.

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

Discussion

Our analyses show that there are large areas in the KBA network, and a large number of species dependent on them, that are not in PAs recognised by the state and where either Indigenous Peoples have rights recognised 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 protected areas is lower on parts of KBAs inside IPL than outside mapped IPL corroborates other studies showing the important role of Indigenous Peoples in reducing tree cover loss (Sze et al., 2021; Pratzer et al., 2023). The wide variation in patterns of tree cover loss between 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 within KBAs in IPLs 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 (O'Bryan et al., 2020; Estrada et al., 2022) and is highly dependent on Indigenous Peoples' stewardship practices, knowledge systems and cultural connections to their lands (Brondizio et al., 2022). Many studies have shown that there is a myriad of contextual factors that influence the ability of IPL to buffer against forest loss, such as land tenure security (Baragwanath and Bayi, 2020), Indigenous knowledge maintenance and revitalization (Fernández-Llamazares et al., 2020), or strong Indigenous-led governance (Artelle et al., 2019), among others. The wide variation found in the rates of tree cover loss in IPLs 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 (Carneiro da Cunha and de Almeida, 2000; Buchadas et al., 2023). While Indigenous communities are pro-actively combatting forest loss in many IPLs through their millennia-old stewardship systems and cultural practices (e.g., Mistry et al., 2016), in other contexts IPLs have been found to be 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 jeopardise the conservation values of such lands, while exacerbating legacies of land dispossession (Farrell et al., 2022; 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 half of the land inside KBAs and mapped IPL is currently inside PAs. The global analysis indicates that globally the tree cover loss in KBAs has been lowest in PAs, regardless of whether they are inside or outside mapped IPL. While 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 both inside IPL and outside mapped IPL before matching, five had higher deforestation rates inside than outside PAs and in eight 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 (West et al., 2006; Schleicher et al., 2019). This may also reflect different PA concepts between countries (e.g. PAs in the UK are very different to those in Brazil), and so their effectiveness varies between (and within) countries too (Geldmann et al., 2014; Cazzolla Gatti et al., 2023).

Notwithstanding deficiencies in the 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 within areas formally allocated to conservation (e.g. dos Santos et al., 2022; Siqueira-Gay et al., 2020; Siqueira-Gay & Sánchez, 2021; Urzedo & Chatterjee, 2021). In such cases, Indigenous Peoples are often at the forefront of resistance and the last line of defence against environmental degradation (Armstrong & Brown, 2019; Spice, 2018) despite suffering higher levels of arrest, violence and even death than other environmental defenders (Scheidel et al., 2020; Beattie et al., 2023). 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 forest may recover its 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).

While the balance of these reasons will vary from case to case, the lack of recognition, respect and/or enforcement of Indigenous Peoples' rights is usually the most powerful driver of deforestation (Baragwanath and Bayi, 2020). This could be one reason behind the pattern observed 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 of mapped IPL). From the advent of colonialism up to the present, Indigenous Peoples have had a history of disempowerment within the 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; Nicaragua, Bryan, 2019; 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 invasions might also lead to deforestation. Indeed, the fact 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, violated and trampled over (International Work Group on Indigenous Affairs, 2022).

Given the pressures on forests, the extent to which IPL will retain forest habitat in future depends on multiple factors. These include: (i) the extent to which Indigenous Peoples' aspirations for the lands they control accord with conservation aims; (ii) the extent to which biodiversity values are retained as a direct consequence of Indigenous Peoples' stewardship and governance of those lands; and (iii) in locations where the influence of Indigenous Peoples' management has been eroded, the extent to which governments are willing both 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). While retention of forest that benefits 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.

Secondly, as noted, 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, while the post-2020 Global Biodiversity Framework's aspirations include "recognising the rights of Indigenous Peoples and local communities, including over their traditional territories..." (CBD, 2022a), the rights of Indigenous Peoples are formally recognised 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 if KBAs on IPL that are currently outside PAs would be best conserved through recognising OECMs, designating new (or expanding existing) PAs, or by recognizing the integrity and distinct nature of IPLs beyond categories of PA and OECMs (as often called for by Indigenous Peoples' organizations; Cariño and Farhan Ferrari, 2021): the answer will be highly context-specific. Whatever the governance system, it is critical that areas should only 'count' towards 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. The analysis also fails to consider historic tree cover loss and the previous influence of IPL, as data were unavailable before 2000. The tree cover dataset 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 do not consider forest regrowth/restoration, and so gain in tree cover, due to the lack of an equivalent reliable tree cover gain layer over the same period that excludes 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 also led to exclusion of many areas within KBAs, and so many grid cells were excluded, as either they could not be statistically matched to equivalent sites outside the IPL boundaries and/or did not have a large enough sample size after matching. This also meant that some countries had lower sample sizes, meaning many countries analysed at the global scale could not be analysed at a national scale. After the matching and the modelling there was considerable spatial autocorrelation in the residuals of the model. We did attempt to control for this by the inclusion of KBA identity (a unique ID for each KBA) as a random effect in the model as an attempt to control for non-independence, but substantial autocorrelations remained in the residuals. This residual non-independence of proximal data points means that the results need to be treated with some caution, as spatial autocorrelation can potentially affect the estimates of site conservation effectiveness (Negret et al., 2020).

The IPL dataset is also incomplete, and there is uncertainty in the exact boundaries of the IPL; see Garnett et al. (2018) for further details on the limitations of this dataset. We acknowledge that blank areas in the IPL dataset do not necessarily reflect a lack of Indigenous Peoples or their lands, but rather represent areas for which ownership or management by Indigenous Peoples cannot be inferred based on the publicly accessible geospatial data compiled in Garnett et al. (2018). The KBA and PA datasets are 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 both KBAs and PAs may also include spatial errors, although marginal boundary issues should not have confounded the analysis because we only included areas that fell wholly within or outside KBAs, IPL and PAs.

Further work

It is critically important that further work explores why tree cover loss (particularly in KBAs) has been lower in some IPL than others, in order to determine if this a result of differences in Indigenous Peoples' rights (including land tenure security), recognition by government, corruption, past history of tree cover loss elsewhere, changes in traditional stewardship practices, land grabs and encroachment, and/or direct deforestation by Indigenous Peoples (and if so why). This also needs to be explored in non-forest systems to understand wider impacts on other habitats and species. This will help us 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 recognised and enforced.

Additionally, our study focussed on land cover change within IPL. Leakage, whereby negative activities such as tree cover loss that would otherwise have occurred within a site are displaced to another location that is unmanaged or protected, can be in issue in estimating site-based conservation effectiveness (Ewers and 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 an issue 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 may be found in the online version of the article at the publisher's website.

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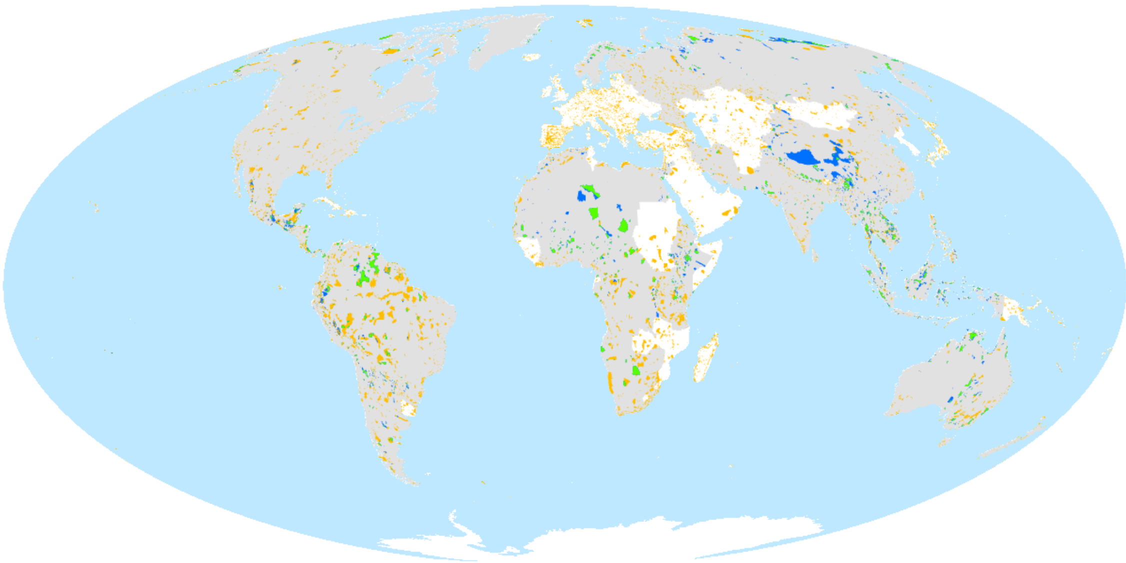

Figure 1: Global distribution of terrestrial Key Biodiversity Areas (KBAs): (1) inside Indigenous Peoples' lands (IPL) and outside Protected Areas (PAs; blue), (2) inside IPL and inside PA (green) and (3) outside mapped IPL (orange). Grey shading of countries indicates countries where IPL are reported, whilst white shading indicates countries are where no IPL were reported or data are unavailable. Note that KBAs with less than 2% of their area overlapping with IPL and/or PAs are treated as not covered by them.

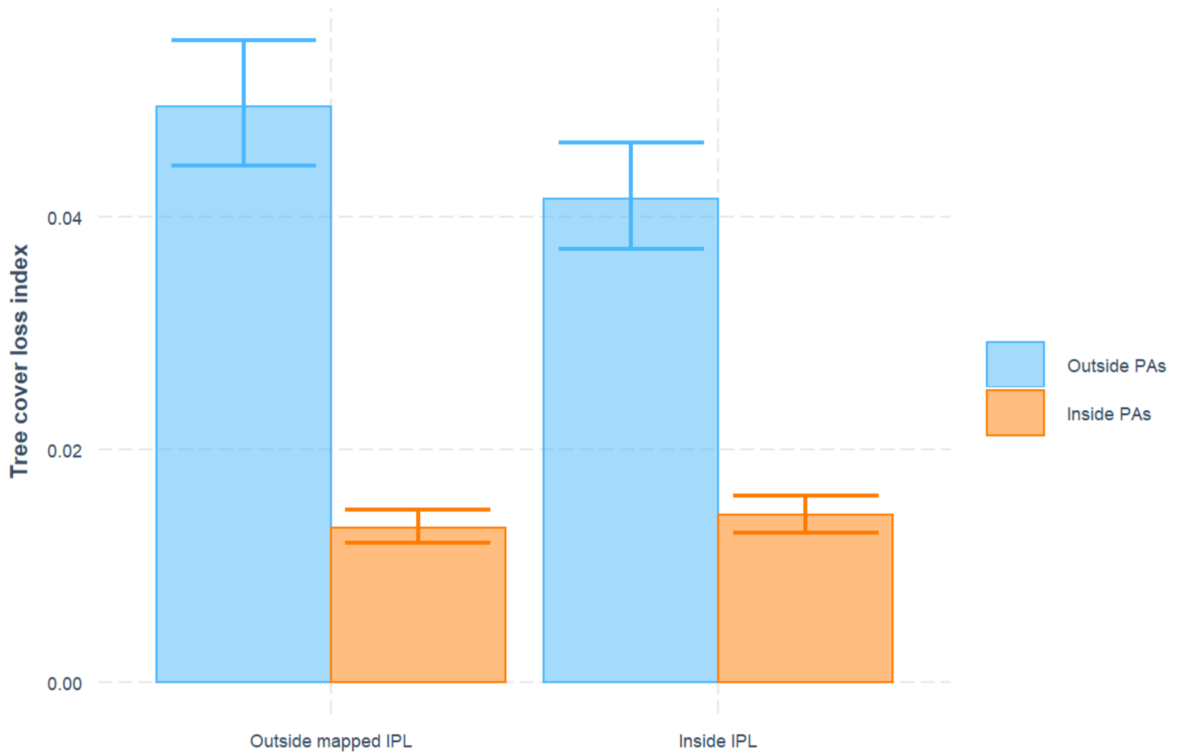

Figure 2. Comparison of tree cover loss rate in Key Biodiversity Areas inside and outside mapped Indigenous Peoples' lands (IPL), and inside or outside Protected Areas (PAs). Bars show a global interaction plot of the outputs from a negative binomial model across countries where IPL were identified by Garnett *et al.* (2018) where KBAs were present following statistical matching (Table 1). The error bars show the 95% confidence intervals.

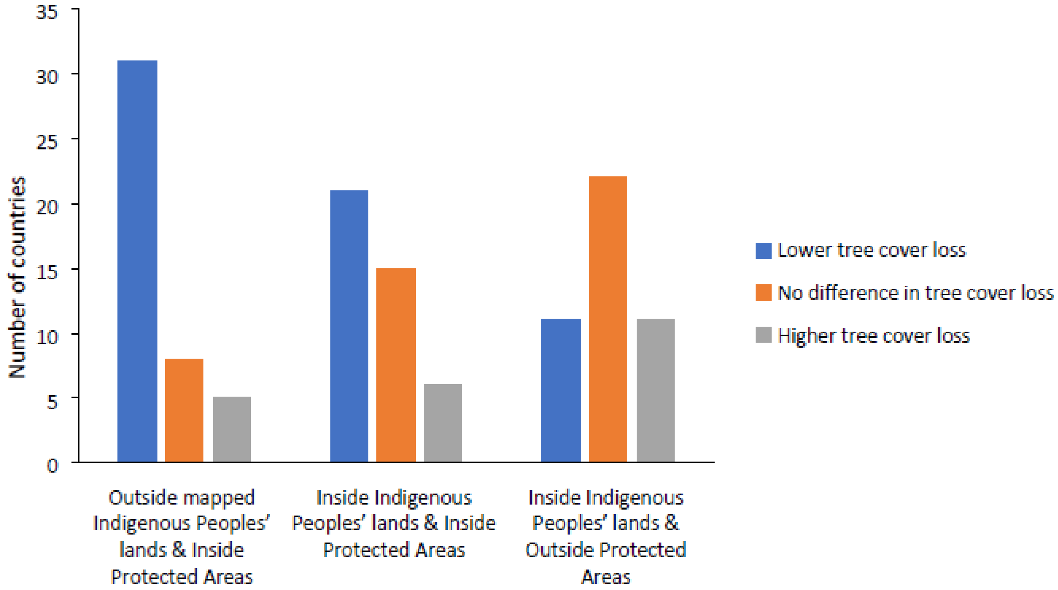

Figure 3: Number of countries that have statistically significantly higher/lower (or no significant difference in the) levels of tree cover loss in their Key Biodiversity Area (KBA) network overall inside Indigenous Peoples' lands (IPL), Protected Areas (PAs) or both compared with KBAs in outside both mapped IPL and PAs. Appendix S5 and Appendix S6 give a full breakdown of the results. The total number of countries was 44, except for two 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.

Table 1. Summary of negative binomial mixed-effect model of tree cover loss between 2001 and 2019 on 1-km^2 cells in Key Biodiversity Areas across all countries where Indigenous Peoples' lands (IPL) were identified by Garnett *et al.* (2018). Variables in italics are continuous, the rest are factors. Significance of Z-tests: P<0.001 ***, P<0.01 **, P<0.05 *, P>0.05 NS.

Variable	**Estimate (SE)**	**Z**

Intercept	-3.92 (0.06)	-70.70***
Inside IPL	-0.17 (0.02)	-9.89***
Inside Protected Area (PA)	-1.31 (0.02)	-75.34***
Inside IPL*Inside PA	0.25 (0.02)	11.61***
Mean tree cover 2000 in 5 km buffer	-0.62 (0.01)	-48.67***
Mean accessibility	-0.73 (0.01)	-86.48***
Mean elevation	-0.37 (0.01)	-47.46***
Mean slope	-0.09 (0.01)	-13.61***
Mean tree cover 2000	-0.06 (0.01)	-7.72***
Shrub Cover, closed-open, deciduous	-0.06 (0.06)	-1.02 NS
Shrub Cover, closed-open, evergreen	0.26 (0.05)	5.56***
Tree Cover, broadleaved, deciduous, closed	-0.51 (0.05)	-11.04***
Tree Cover, broadleaved, deciduous, open	-0.55 (0.07)	-7.78***
Tree Cover, broadleaved, evergreen	-0.14 (0.04)	-4.00***
Tree Cover, mixed leaf type	-0.3 (0.09)	-3.38***

Tree Cover, needle-leaved, deciduous	-2.51 (1.37)	-1.83 NS
Tree Cover, needle-leaved, evergreen	-0.16 (0.06)	-2.64**
Tree Cover, regularly flooded, fresh water	-0.52 (0.04)	-11.68***
Tree Cover, regularly flooded, saline water	-0.26 (0.13)	-1.96*