Quantifying conservation outcomes in Indigenous peoples' lands across the tropics

J. S. Sze

A thesis submitted in partial fulfilment of the requirements for the degree of Doctoral of Philosophy.

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General Summary

Biodiversity loss and climate change represent some of the biggest challenges humanity currently faces, with habitat loss as the biggest proximate driver. Area-based conservation is a key conservation policy and recent international conservation targets aimed for at least 30% of terrestrial, inland water, and of coastal and marine areas to be effectively conserved and managed through protected areas and other effective area-based conservation measures, recognising Indigenous and traditional territories, by 2030. While the effectiveness of protected areas (PAs) in achieving conservation outcomes has received much research attention in recent years, there still remains a gap in a global-scale understanding for Indigenous lands (ILs). Focusing on tropical forests, as globally important biomes for biodiversity and climate change mitigation, this thesis quantifies three metrics of conservation outcomes on ILs, PAs, the spatial overlap of protected areas and Indigenous lands (PIAs), and non-protected areas across the tropical Americas, Africa, and Asia. In Chapter 2, I examined deforestation and forest degradation rates from 2011-2019 using propensity score matching and generalised linear mixed models. I found that deforestation was reduced by 16.8-25.9% and degradation reduced by 9.1-18.4% on ILs compared to non-protected areas across tropical regions, while differences compared to PAs varied between regions. In Chapter 3, I sought to investigate forest integrity using the Forest Landscape Integrity Index which incorporates observed pressures, inferred pressures, and lost connectivity, and long-term human land-use intensity using the Anthromes dataset. Across tropical regions, forest integrity was highest and land-use intensity the least in PIAs, but varied in ILs between regions compared to non-protected areas. In Chapter 4, I assessed 11,872 forest-dependent vertebrate species' Area of Habitat and compared species richness, extinction vulnerability, and range-size rarity inside and outside Indigenous peoples' lands. At least 76.8% of tropical amphibians, birds, mammals, and reptiles had range overlaps with ILs, with an average range overlap of ~25%. Most countries in the Americas had higher species richness in ILs than outside, whereas most countries in Asia had lower extinction vulnerability scores in ILs, and more countries in Africa and Asia had slightly higher range-size rarity in ILs. Taken together, the thesis reveals the contributions that Indigenous peoples' lands make towards tropical conservation, in terms of reducing habitat loss, maintaining habitat quality, and providing vital habitat for forest-dependent vertebrate diversity. Supporting and including Indigenous peoples in conservation target-setting and planning is not only socially just, it is likely vital to the success of achieving the Kunming-Montreal targets.

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Declaration

I confirm that the Thesis is my own work. I am aware of the University's Guidance on the Use of Unfair Means. This has not previously been presented for an award at this, or any other, university.

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Chapter 1

General Introduction

1.1. The biodiversity and climate crisis

The Earth is undergoing profound transformations in its geological history, with sufficiently deep transformations in Earth systems planetary functioning and biosphere that geologists are considering a new geological epoch for our present era – the Anthropocene (Malhi, 2017). Globally, we are seeing average vertebrate species losses 100 times higher than the background rate of 2 extinctions per million species per year in the last century (Ceballos et al., 2015).

These high levels of extinction, considered the 6th major extinction event on Earth, are driven by habitat and land-use change, overexploitation of natural resources, pollution, climate change, and invasive species (Jaureguiberry et al., 2022; MEA, 2005). With population and abundance declines across most species groups and range shrinkages (Ceballos et al., 2017; WWF, 2022), the decline in biodiversity is affecting ecosystem functioning and loss of ecosystem services ranging from pollination and pest control to nutrient cycling, water regulation, human health and food security (Dirzo et al., 2014). It has been estimated that the proposed planetary boundary for biodiversity loss within which ecosystem function is relatively unaffected has been exceeded (Newbold et al., 2016).

Of the proximate drivers of biodiversity loss, habitat loss, in particular conversion to monoculture agriculture (Hoang et al., 2023), has an outsized impact. Alongside fossil fuel extraction and use, habitat loss contributes to greenhouse gas emissions and climate change (Kastner et al., 2021). The loss of habitat connectivity also prevents species from adapting to climate change through range shifts (Senior et al., 2019), furthering species' extinction risks and their declines (Spooner et al., 2018), while climate change and global warming itself contributes to further habitat loss and fragmentation (Segan et al., 2016). These twinned biodiversity and climate crises thus arguably represent the biggest threats to the socio-ecological and political economy of our present human society, with Indigenous and natural resource-dependent communities being disproportionately affected (Ford et al., 2020).

It is increasingly acknowledged that the proximate drivers of the biodiversity and climate crisis are ultimately a result of neoliberalised capitalism that relies on continuous growth and extraction of material and value, resulting in regional trade agreements, land investments and land-grabbing (Abman & Lundberg, 2020; Ceddia, 2020; Moranta et al., 2021; Tulone et al., 2022) and the growing affluence of urban and globalised populations (Marques et al., 2019; Weinzettel et al., 2013; Wiedmann et al., 2020). Nonetheless, to address the proximate drivers of biodiversity decline, international conservation policies, such as the Convention on Biological Diversity (CBD) that first convened at the Earth Summit in Rio de Janerio, Brazil in 1992 and was adopted the following year, focus on area-based measures (Maxwell et al., 2020; Watson et al., 2014). Area-based conservation measures, primarily thus far in the form of Protected Areas (PAs), are a cornerstone of conservation policy. PAs are defined as "a clearly defined geographical space, recognized, dedicated and managed through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values", and presently, terrestrial PAs cover just under 16% of the Earth's surface (UNEP-WCMC & IUCN, 2023), though the recent Kunming-Montreal post-2020 Global Biodiversity Framework laid out a target for at least 30% of terrestrial and inland waters to be under protection by 2030 (CBD, 2022).

Although area-based conservation has largely been found to be effective in achieving conservation outcomes (Section 1.3), strict protected areas have also been plagued by injustices to Indigenous and local communities living in or around conservation areas (Section 1.2.1). International conservation policies have thus shifted in their language and framings towards acknowledging historic and current injustices, recognising the contributions made by Indigenous communities, and requiring their inclusion in decision-making processes (Section 1.2.2). These contributions fall largely within the concept of Other Effective area-based Conservation Measures (OECMs), introduced in 2010 to the Aichi Biodiversity Targets for 2020 (CBD, 2010), though an official definition was not provided until 2018. With the addition of OECMs, areas which may not have an explicit conservation objective, but which have demonstrated conservation outcomes, are included within national area targets. Such areas include religious or sacred sites and areas within Indigenous or local communities' territories managed through customary institutions and regulations (Kothari et al., 2013). Additionally, in the recent Kunming-Montreal post-2020 Global Biodiversity Framework adopted last December, the target for 30% included PAs, OECMs, and the recognition of Indigenous territories as separate to OECMs (CBD, 2022).

This shift in policy narratives is accompanied by greater emphasis on the evaluation of effectiveness of conservation policies (Ferraro & Pattanayak, 2006), including of PAs and Indigenous territories (Gurney et al., 2023). Yet while there have been local and regional studies, particularly in the Amazon, there remains a gap in understanding of how Indigenous peoples' lands contribute to conservation at the pan-tropical scale. This thesis thus sets out to fill this gap in knowledge by quantifying how Indigenous peoples' lands are associated with deforestation and degradation, forest integrity, and forest-dependent vertebrate species across the tropics.

The following sections in this chapter outline the history of area-based conservation measures, followed by injustices enacted on Indigenous peoples as a result of conservation policies, and the move towards inclusive conservation in international policies. I then discuss the effectiveness of area-based conservation measures, focusing on tropical forests as one of the most important biomes, which sets out the rationale for this thesis. I conclude with an overview of the rest of the chapters and a short reflection on my positionality.

1.2. History of area-based conservation measures and their impacts on Indigenous peoples

1.2.1. Growth of PAs and 'fortress conservation'

Human communities have long used and managed areas for a wide variety of purposes, for subsistence, economic livelihoods, cultural and spiritual practices, and maintenance of ecosystem services. These management practices have included setting aside certain areas within their territories as sacred spaces, for watershed protection, or for future generations' uses (Berkes, 2017; Colding & Folke, 2001; ICCA Consortium, 2021; Kemf, 1993).

Modern area-based conservation measures primarily in the form of PAs originated in the late 19th century, beginning with Yellowstone National Park in the United States of America in 1872. Yellowstone National Park was established amidst wars against the Indigenous North Americans to subdue and subjugate them under American control. The Shoshone, Lakota, Crow, Bannock, Nez Perce, Flathead, and Blackfeet peoples were forcibly removed from within the park boundaries and forbidden from using those lands, resulting in their resistance and mass killings (Kemf, 1993).

Such strict PA concept demarcated areas that were set aside primarily for conservation of particular charismatic wildlife or iconic landscapes and managed by the State to the exclusion of Indigenous and local residents and their use, commonly termed as 'fortress conservation'. The PAs were established often without the Free, Prior, and Informed Consent (FPIC) of communities living in these newly designated spaces, their participation in decision-making, or even awareness, until the day of eviction (Brockington & Igoe, 2006; Colchester, 2004). This model of PAs spread from North America to the rest of the world, where European colonial powers designated large areas of their colonies as wildlife reserves or National Parks (Gurney et al., 2023; Watson et al., 2014). In cases where Indigenous communities were not evicted to make way for newly created PAs, there were often resource use and access restrictions imposed, threatening the food security, livelihoods, and cultural and spiritual identity of these communities (Adams & Mulligan, 2003; Colchester, 2004; Tauli-Corpuz et al., 2020). These resulted in increased conflicts, including killings, sexual violence, and intimidation, between Indigenous communities and State forces, particularly when armed military or para-military forces were sent in to secure and guard PAs and their associated biodiversity (Domínguez & Luoma, 2020; Kashwan et al., 2021; Tauli-Corpuz et al., 2020).

Such 'fortress conservation' approaches are rooted in European Enlightenment thinking of universality and objectivity that imposed the separation of humans and nature (Adams & Mulligan, 2003; Fletcher et al., 2021). This 'human-nature' duality was writ large particularly by North American colonists' ideas of wilderness and their perception that nature needed to be protected from humans. Nature thus envisioned needed to be kept uninhabited, to be used for recreation and science, but otherwise undisturbed (Robbins, 2007). Such ideas of 'pristine wilderness' were imposed particularly on African landscapes, for example through the creation of the first National Park in British colonial Tanganyika (now Tanzania), the Serengeti National Park in 1948, which only allowed the Maasai, Ndorobo, and Sukuma peoples to stay if they remained "primitive" (Neumann, 2003).

From just under 170 PAs established by 1900, the number of terrestrial and inland waters PAs grew rapidly post World War II. As European colonial powers granted independence to their former colonies, the establishment of PAs increased. This was facilitated by international conservation agreements, global conservation organisations, and multilateral and bilateral funding institutions (Zimmerer et al., 2004). At present, there are 266,984 terrestrial and inland waters PAs (and an additional 634 OECMs) in nearly every country (Figure 1; UNEP-WCMC & IUCN, 2023).

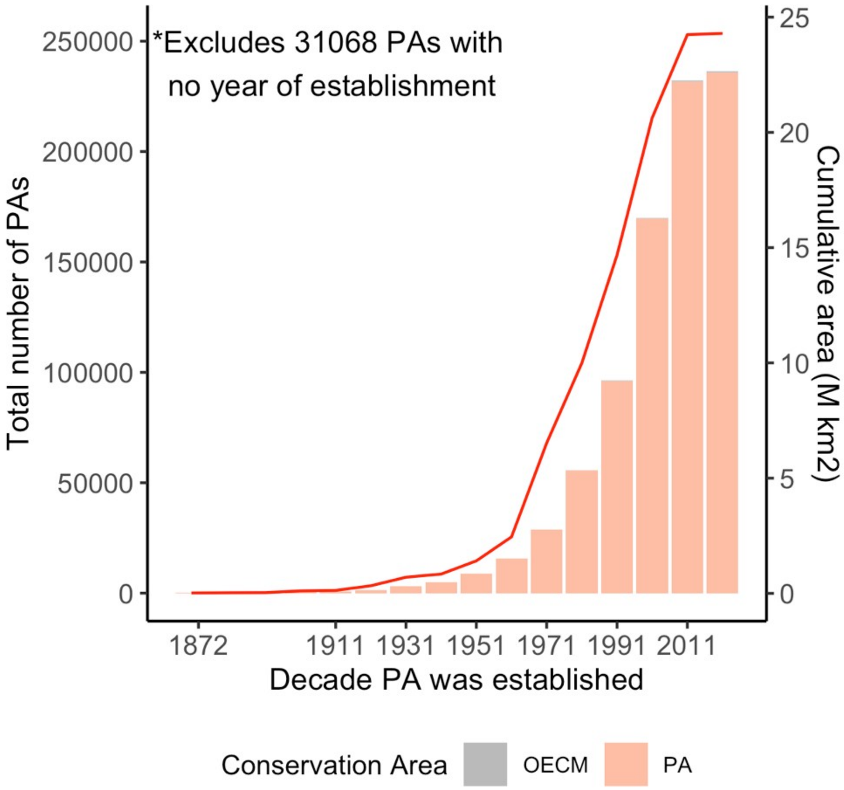

Figure 1. Cumulative number of protected areas (PAs) and cumulative area covered (red line) up to May 2023. Data from protectedplanet.net, accessed 5th May 2023.

Alongside the growth of PAs, there was increasing recognition of the social implications of PAs, particularly the impact they had on displacing Indigenous peoples (Adams & Mulligan, 2003; Brockington & Igoe, 2006; Colchester, 2004). Where data was available, evictions for PAs were found to be concentrated in Africa, South and Southeast Asia, and North America (Brockington & Igoe, 2006), reflected by a shift towards strict PAs in these regions (Zimmerer et al., 2004). The continuation of European colonial and capitalist logics of individual land ownership and making land productive and economically profitable by post-colonial States perpetuated the dispossession of Indigenous communities (Domínguez & Luoma, 2020; Kashwan et al., 2021). Additionally, State logics of exerting territorial sovereignty came into play, with post-colonial States using the creation of PAs, especially transboundary PAs, to bring remote spaces and peoples under their control (Lunstrum, 2013; Peluso, 1993; Peluso & Vandergeest, 2001). In tandem with needing to make these spaces profitable through encouraging nature-based tourism, State governments wanting to replicate colonial ideas of 'wilderness' have evicted Indigenous peoples from these PAs (Neumann, 2003).

Yet, while these eviction trends are true across tropical Africa and Asia, most Latin American PAs that were created accepted the presence of Indigenous peoples, though in some cases there was still the caveat that they had to maintain their traditional subsistence practices (Davis & Wali, 1994). Some of these PAs were created with the aim of protecting Indigenous peoples, some of whom were uncontacted, and biodiversity, such as Xingu National Park in Brazil and Manu National Park in Peru (Davis & Wali, 1994). In other cases, Indigenous peoples requested State governments and international NGOs to create parks and PAs that would deter external colonisers and resource extraction activities, for example the Shuar in Ecuador's Kutuku Protected Forest (Rudel, 2009). The Latin American model of PAs, although still entwined with ideas of enforcing State territories, were more accepting of human communities within PAs and from the 1990s, shifted towards recognising the rights of Indigenous peoples in their respective constitutions, including rights to their territory (Zimmerer, 2011).

1.2.2. Move towards inclusion of Indigenous peoples in international conservation policies

The International Union for Conservation of Nature and Natural Resources (IUCN) was the first global conservation organisation established in 1948 to coordinate conservation activities across State governments and civil society. In 1958, the IUCN established what is now the World Commission on PAs, and record-keeping of established PAs started in 1962. At the World Parks Congress in 1975, the Kinshasa Resolution acknowledged that Indigenous Peoples should not be displaced from their traditional lands, recognising their traditional ways of living and land ownership (Colchester, 2004). In 1982, at the 3rd World Parks Congress in Bali, it was reaffirmed that Indigenous communities had the right to social, economic, cultural, and spiritual self-determination, and the right to participate in decision-making processes affecting their lands and natural resources. The 3rd World Parks Congress also saw the first proposal for an area-based target of 10% of the Earth's land surface to be in PAs. This target of at least 10% of each biome to be within PAs was subsequently established at the 4th World Parks Congress in Caracas in 1992, to be achieved by 2000.

This move towards redressing past injustices and creating new policies and practices that are more inclusive came alongside narratives that conservation should do more than protect the environment and wildlife; it should also achieve social and economic objectives (Adams & Hutton, 2007). Many communities living around conservation areas were often more dependent on natural resources and defined as economically poor by States, and conservation initiatives were perceived to need to promote their development. There was thus a boom in Integrated Conservation and Development

Projects and Community-Based Natural Resources Management in the 1980s and 1990s, focusing on the more equitable distribution of benefits and greater participation in decision-making processes (Adams & Hutton, 2007; Martin et al., 2013). This was reflected in the first guidelines published by the IUCN in 1994 classifying PAs into six categories following their management objectives, ranging from strict protection to managed resource extraction (IUCN, 1994). The emphasis on greater participatory or community-based conservation, however, was limited to management; PAs were often still planned, initiated, and governed by States.

In 1989, the International Labour Organisation adopted Convention 169 Concerning Indigenous and Tribal Peoples in Independent Countries (ILO 169). It was the first and is still the only international legally-binding policy instrument to recognise Indigenous peoples' relationships and rights to the lands they traditionally occupy and the associated natural resources. Given the diversity of Indigenous peoples worldwide, a formal universal definition is not prescribed, rather Indigenous peoples are identified primarily by self-identification: self-identification as Indigenous peoples at the individual level and accepted by the community as their member. Other criteria by which Indigenous peoples are identified include being descended from populations who have inhabited the geographical region at the time of conquest, colonisation, or establishment of present State boundaries, and having retained some or all of their own social, economic, cultural, and political institutions, distinct from the dominant national community (ILO, 1989).

This was followed by the development of United Nations' Declaration on the Rights of Indigenous Peoples (UNDRIP) in 1993, although it was not adopted by the United Nations General Assembly until 2007. Although the UNDRIP is not legally binding, it represents the minimum standards for the protection of the rights and fundamental freedoms of Indigenous peoples, including their rights to non-discrimination, social development and well-being, self-determination, political representation, land and resource tenure, FPIC regarding their lands and territories, and the enjoyment of their customary institutions (United Nations, 2007). These international policy developments were mirrored in the conservation sphere with the adoption of several Resolutions regarding Indigenous peoples at the 1996 Montreal World Conservation Congress. Resolution 53, specific to PAs, recognised the rights of Indigenous peoples to their territory and resources within PAs, rights to be consulted and participate in effective decision-making processes, as well as the need to reach agreement regarding PAs in their lands (IUCN, 1996b). A further Resolution 51 recommended member countries to adopt and implement ILO 169 and comply with UNDRIP (IUCN, 1996a).

At the 5th World Parks Congress in 2003, the Durban Accord went beyond recognising their rights to explicitly include restitution of their rights in existing parks and their inclusion when establishing and managing PAs. This was achieved in part through the strong advocacy of Indigenous peoples and presence at the Congress (Colchester, 2004). The 4th World Conservation Congress in 2008 in Barcelona saw a fundamental shift in paradigm, moving beyond just management to incorporate governance (Kothari et al., 2013). While management refers to operational decisions to achieve specific outcomes, governance expands to the broader processes and institutions involved in decision-making, such as who gets to participate and what decisions are made (Armitage et al., 2012). These include regulatory processes, mechanisms, and organisations through which various actors influence environmental actions and outcomes (Lemos & Agrawal, 2006). The revised PA management guidelines in 2008 kept the six management categories from I (Strict Nature Reserve) to VI (Protected Area with Sustainable Use of Natural Resources), but now included four main governance types (Figure 2; Dudley, 2008). Up to that point, it was largely assumed that only the State was able to manage or govern PAs; this had now expanded to Indigenous peoples and local communities or private actors. These shifts in paradigm towards decentralised governance approaches reflected the increased recognition that Indigenous and local communities, whose lands now fell within PA boundaries, had the right to partake in decision-making, and the importance of good governance, entailing aspects of accountability, equitability and inclusivity, effectiveness, efficiency, legitimacy, transparency, and participatory consensus-derived decision-making (Armitage et al., 2012).

Governance types / Protected area categories	A. Governance by government			B. Shared governance			C. Private governance			D. Governance by indigenous peoples and local communities	
	Federal or national ministry or agency in charge	Sub-national ministry or agency in charge	Government-delegated management (e.g., to an NGO)	Transboundary management	Collaborative management (various forms of pluralist influence)	Joint management (pluralist management board)	Declared and run by individual land-owners	… by non-profit organizations (e.g., NGOs, universities)	… by for-profit organizations (e.g., corporate owners, cooperatives)	Indigenous peoples' protected areas and territories – established and run by indigenous peoples	Community conserved areas – declared and run by local communities
Ia. Strict Nature Reserve											
Ib. Wilderness Area											
II. National Park											
III. Natural Monument											
IV. Habitat/ Species Management											
V. Protected Landscape/ Seascape											
VI. Protected Area with Sustainable Use of Natural Resources											

Figure 2. The IUCN protected area matrix: a classification system for protected areas comprising both management category and governance type. Reproduced from Dudley (2008).

This reflection that respecting of human rights and conservation interventions can go hand-in-hand is demonstrated by the formation of the Conservation Initiative on Human Rights (CIHR) in 2009, a group of eight largest international conservation organisations including Birdlife International, Conservation International, Fauna & Flora International, IUCN, The Nature Conservancy, Wetlands International, Wildlife Conservation Society, and World Wildlife Fund for Nature. Participating organisations of the CIHR committed to uphold a set of human rights principles and to implement policies and appropriate accountability mechanisms (CIHR, 2014). Since 2014, the IUCN has also developed a Standard on Indigenous Peoples that establishes risk assessments and management requirements for IUCN projects where Indigenous Peoples are involved, as part of IUCN's Environmental and Social Management System (IUCN, 2019).

In 2010, the CBD adopted the Aichi Biodiversity Targets, of which Target 11 was relevant to area-based conservation. Apart from setting an area-based target of 17% of the Earth's land surface to be protected by 2020, Target 11 also included provisions for areas of biodiversity importance, ecological connectivity and representativeness, as well as achieving equity and effectiveness. Thus, alongside PAs, Target 11 included the category of Other Effective area-based Conservation Measures (OECMs), which can include territories and areas conserved by Indigenous peoples (CBD, 2010). Although a formal definition of OECMs was not adopted until 2018, it was largely envisaged that OECMs would support recognition of Indigenous peoples as rightful custodians and equal partners in conservation (Jonas et al., 2014). It is also increasingly acknowledged that without the equitable participation of Indigenous peoples and recognition of their territory, conservation targets would not be met (Dudley et al., 2018; Gurney et al., 2023).

This is further emphasised in the embedding of a rights-based approach in the most recent Kunming-Montreal post-2020 Global Biodiversity Framework, and with the explicit acknowledgement of rights of Indigenous peoples and their territory within Target 3, relating to area-based conservation (CBD, 2022). The Intergovernmental Platform for Biodiversity and Ecosystem Services (IPBES) had also promoted the incorporation of plural values of nature, recognising that Indigenous and traditional knowledges have often been ignored or side-lined as inferior to that of modern Western science (IPBES, 2019). These changes in policy reflect growing acknowledgement and recognition that Indigenous peoples and local communities have managed their land for generations in ways that can be beneficial to biodiversity conservation, and their different understandings of nature and worldviews are often integral to their relationships with the environment. They are also supported by evidence that where Indigenous and local community are empowered to govern and manage conservation areas and are supported by national legislation and policies conservation, positive well-being and ecological outcomes are more likely, enabling equitable and effective conservation (Dawson et al., 2021).

Despite progress being made at international policy level, implementation by nation states is variable across regions. A global database of environmental conflicts found that 4% of cases (n=2743) reported between 2011 and March 2019 were over conservation, mostly in low-income countries (Scheidel et al., 2020). Many countries in sub-Saharan Africa have continued with 'fortress conservation' approaches, such as plans to forcibly evict 150,000 Maasai peoples from their ancestral lands in Ngorongoro Conservation Area and Loliondo in Tanzania (UN News, 2022). Similarly in Asia, rights violations against the Karen peoples have been reported in Kaeng Krachan Forest Complex in Thailand (OHCHR, 2021). These concerns are exacerbated by the fact that most countries in these regions have yet to ratify ILO 169 or adopt UNDRIP, and secure land tenure is still lacking for many Indigenous communities (RRI, 2020). In the Americas, while Indigenous peoples in most countries have constitutional recognition of their territorial rights, there still exists limitations to exercising those rights and the suppression of Indigenous organisations and governments, though most conflicts in the region relate to extractive and development projects rather than conservation (ECLAC, 2014).

At the same time, there has been some acknowledged successes in countries such as Namibia and South Africa, where there has been greater emphasis on co-management of conservation areas and recognition of traditional knowledges and respect for customary institutions (Armitage et al., 2020). Indonesia had also recognised ownership of customary forests in a 2012 Constitutional Court decree, resulting in the restitution of 18 Customary Forest titles from State Forests in 2016 and 2017 (Hidayat et al., 2018). These examples demonstrate the small stepping stones towards actually implementing the full suite of rights of Indigenous peoples in conservation.

1.3. Effectiveness of area-based conservation measures

1.3.1. Early evaluations using surveys and inside-outside comparisons

Alongside the move towards inclusive and just conservation, there have also been calls to evaluate the effectiveness of protected areas in achieving conservation objectives. Concerns over the proliferation of 'paper parks', PAs that were quickly established but without effective management, arose in the 1990s (Brandon et al., 1998). One of the earliest evaluations of the effectiveness of PAs relied on an expert survey of management status and threats of 46 PAs in 10 highly forested countries that the World Bank operated in (Stolton & Dudley, 1999). Bruner et al. (2001) conducted a pan-tropical evaluation across 22 countries using questionnaires to assess anthropogenic impacts on 93 PAs, while a subsequent survey led by World Wide Fund for Nature (WWF) in >200 PAs from 37 countries utilised a tracking tool to assess management effectiveness and threats (Dudley et al., 2004). These studies found that PAs face many threats and were not perceived to be very effective, although the majority of the PAs surveyed were perceived to reduce deforestation (Bruner et al., 2001).

Without undermining the value of surveys and questionnaires, these studies provided an understanding of whether PAs were perceived by PA managers to be effective. However, in the few decades since the launch of the satellite Landsat 1 in 1972, the increasing availability of satellite images allowed for studies combining satellite-derived data with field-based surveys to evaluate the effectiveness of PAs in actually achieving conservation outcomes. Compiling results from 20 studies of 49 PAs across the tropics, Naughton-Treves et al. (2005) found that of the 36 PAs where authors compared deforestation rates inside and outside PAs, majority of the PAs had lower deforestation rates. Similarly, a systematic review found that 62 of 76 studies reported higher habitat loss outside PAs than inside (Geldmann et al., 2013).

With the advent of improved computing power, the publication and availability of global wall-to-wall satellite-derived tree cover and tree cover loss data (Global Forest Watch; Hansen et al., 2013) made it possible to move beyond regional and country analyses to global analyses. Across tropical and subtropical moist forest PAs in 56 countries, most PAs did reduce forest loss between 2000 and 2012 compared to varying distances of buffer zones outside PAs, though not in Asia (Spracklen et al., 2015). With the same Global Forest Watch data but comparing annual forest change for 4028 PAs in 64 countries inside and outside PAs using panel regression with park characteristics, Blankespoor et al. (2017) found that effective protection was positively and significantly associated with park size, national park status, and management by Indigenous peoples.

1.3.2. Evaluations using quasi-experimental methods

Although PAs were found to be generally effective in reducing deforestation relative to their buffer zones outside, it has been argued that PAs should be compared to similar areas (Naughton-Treves et al., 2005). Ferraro & Pattanayak (2006) called for conservation policies to be rigorously assessed using programme evaluation methods that determine which policies work and when, specifically focusing on comparing observed with expected outcomes if interventions were not made (i.e. counterfactual). They advocated for the use of quasi-experimental designs to control for confounding factors, variables that are associated with both the intervention and with the outcome, especially that of endogenous selection of how units are chosen for intervention (Ferraro & Pattanayak, 2006). Quasi-experimental designs thus approximate experimental designs (i.e. randomised controlled trials) in controlling for confounding factors and attributing causality and outcomes to the intervention, useful when it is not financially, practically, legally, politically, or ethically feasible to implement experimental designs. One popular approach to identify control units similar to intervention units are statistical matching methods (Schleicher et al., 2020).

PAs are often located on more economically marginal lands that are steeper, higher in elevation, and further from population centres (Joppa & Pfaff, 2009). Andam et al. (2008) used Mahalanobis matching of land use productivity, distance to forest edge, distance to roads, and distance to major city to measure the effectiveness of Costa Rican PAs, finding that PAs reduced deforestation between 1960 and 1997, but failure to account for confounders overestimated avoided deforestation.

In Sumatra, Indonesia, with the use of propensity score matching of baseline 1990 forest cover, slope, elevation, distance to forest edge, distance to roads, distance to logging roads, and political province, PAs were also found to have reduced deforestation between 1990 and 2000, and similarly, not controlling for neighbourhood leakage and location biases overestimated avoided deforestation (Gaveau et al., 2009).

At a global level, using land cover changes between 2000 and 2005 and Mahalanobis matching of elevation, slope, distance to roads, distance to urban areas, and ecoregions, Joppa & Pfaff (2011) showed that matching reduced estimated changes in land cover by half compared to both naïve estimates (changes in land cover of PAs without a comparator) and 10 km buffer zone estimates. Most recently, using the Global Forest Watch dataset from 2000 to 2018, >18000 terrestrial PAs were evaluated using coarsened exact matching of elevation, slope, tree cover, travel time to nearest densely populated area, population density, country, ecoregion, and primary driver of forest cover loss, finding that PAs reduced but did not eliminate deforestation (Wolf et al., 2021).

1.3.3. Comparing PAs managed by States with Indigenous approaches

PAs are largely acknowledged to be effective in reducing deforestation, however, there has also been an increasing trend of PA Degrading, Downsizing, and Degazettement to allow for extractive industrial mining, concessions, and oil exploration (Golden Kroner et al., 2019; Watson et al., 2014). Alongside the known inequities arising from establishing PAs, it is increasingly necessary to understand of how Indigenous approaches compare with state-managed PAs in achieving conservation outcomes.

One of the earliest comparisons between strict PAs and Indigenous lands was in the Brazilian Amazon, which found that both had reduced deforestation between 1997 and 2000 compared to the 10 km buffer zone outside, but that Indigenous lands were usually located within active agricultural frontiers (D. Nepstad et al., 2006). In the Peruvian Amazon, PAs and Indigenous territories both reduced deforestation between 1999 and 2005 compared to non-protected areas outside, though PAs had much lower deforestation rates (Oliveira et al., 2007). Similarly, PAs and Indigenous reservations in Colombia Guyana Shield reduced deforestation between 1985 and 2002 relative to their 10 km buffer zones, though PAs also had lower deforestation rates (Armenteras et al., 2009).

As awareness around the importance of robust study designs and elimination of confounding factors and location biases grew, studies using quasi-experimental designs also became more common, finding similarly that PAs were more effective than Indigenous lands. In Acre, Brazil, deforestation in strict PAs and Indigenous lands were reduced between 2000 and 2008 when propensity score matched on distance to nearest road, distance to nearest city, distance to forest edge, soil quality, precipitation, and slope compared to counterfactual controls, but strict PAs avoided more deforestation (Pfaff et al., 2014). Similarly, using Mahalanobis matching on elevation, slope, distance to roads, and distance to towns, Vergara-Asenjo & Potvin (2014) found that although Indigenous territories and PAs both reduced deforestation between 1992 and 2008 in Panama compared to counterfactual controls, PAs avoided more deforestation.

However, other studies using quasi-experimental designs have also found that Indigenous lands were more effective than PAs. Across Latin America and the Caribbean, using matching on distance to roads, distance to major cities, elevation, slope, and rainfall, Nelson & Chomitz (2011) found that Indigenous areas avoided twice as many forest fires during 2000 to 2008, as a proxy for deforestation, compared to PAs. Matching on elevation, slope, probability of flooding, baseline forest cover, distance to forest edge, travel time to major cities, and state, Nolte et al. (2013) similarly found that Indigenous lands were usually just as effective as strict PAs in reducing deforestation between 2000 and 2010 in Brazilian Amazon compared to counterfactual controls, with Indigenous lands particularly effective in high deforestation pressure areas. In the Peruvian Amazon, both state PAs and Indigenous territories had significantly lower deforestation and degradation rates from 2006 to 2011 compared to propensity score matched counterfactual controls, but that Indigenous territories were more effective at avoiding deforestation (Schleicher et al., 2017). Using propensity score matching on slope, elevation, flooding, precipitation, distance to nearest deforestation, distance to roads, distance to rivers, distance to cities, latitude and longitude, and state, Alves-Pinto et al. (2022) also found that native vegetation conversion between 2005 and 2017 in Brazilian Amazon was lower in both Indigenous lands and PAs compared to counterfactual controls, with Indigenous lands avoiding much more conversion.

PAs and Indigenous lands thus are both found to have reduced deforestation in general, although whether PAs or Indigenous lands are more effective is variable across time and across countries and regions within countries. However, most of the available studies applying quasi-experimental designs to evaluate the effectiveness of PAs and Indigenous lands have focused on the Americas, since the region has had a longer history of demarcating Indigenous lands and making such spatial data available for research. It is thus still unknown if Indigenous lands in tropical Africa and Asia similarly reduce deforestation, and how they compare with PAs.

1.3.4. Importance of Indigenous lands for faunal diversity

While most studies evaluating conservation outcomes have focused on forest loss since tree cover is more easily derived using satellite images, it is widely acknowledged that retention of forest cover does not necessarily imply protection of wildlife living within the forests; the ‘empty forest’ syndrome (Redford, 1992). Given the logistical and financial difficulties in deploying a global biodiversity survey across PAs, many global studies evaluating the effectiveness of PAs on faunal diversity rely on summaries of available literature.

From a systematic review of 42 studies from 35 papers, Geldmann et al. (2013) found that although more papers had reported positive results, on balance there was inconclusive evidence that PAs were effective in maintaining species populations compared to varying counterfactuals, especially given small sample sizes and geographical and taxonomic biases. A subsequent meta-analysis of 86 studies covering 32 countries and 57 PAs found that PAs had higher species abundance, assemblage abundance, and richness for mammals, birds, herptiles, and arthropods compared to surrounding areas (Coetzee et al., 2014). Using a collated database of abundances and occurrences inside and outside 359 PAs from 156 studies covering 13,669 species of vertebrates, invertebrates, and plants, PAs were found to have higher species richness and abundance than outside (Gray et al., 2016).

While these studies provide a grounded understanding of the effectiveness of PAs and the state of biodiversity within them, another way of evaluating PAs is how much they cover species’ geographic ranges and protect their habitats (Brooks et al., 2004). Improvements in computing power and Geographic Information System technologies have aided the IUCN Red List of Threatened Species assessment of species’ extinction risk, in particular the documentation of species’ geographic ranges. First established in 1964, the IUCN Red List has assessed 150,388 species covering birds, reptiles, amphibians, mammals, insects, fishes, marine invertebrates, and plants, of which >82% have spatial data (IUCN, 2023).

Although these spatial data on species’ extent of occurrence are known to be an overestimate of their true area of occupancy, when combined with information on the species’ habitat and elevational ranges and satellite data on land cover and elevation, their Area of Habitat can be estimated which is more useful in assessing species’ ranges within PAs (Brooks et al., 2019). Such Area of Habitat maps have been used to identify global forests with high value for amphibians, birds, mammals, and conifers (S. L. L. Hill et al., 2019), to assess impact on amphibians, birds, mammals, and plants arising from land-use change in the Brazilian Cerrado (Durán et al., 2020), and to identify areas with high biodiversity and carbon values that are within PAs (Soto-Navarro et al., 2020).

Focusing on the use of species' geographic range data for evaluating species range coverage in Indigenous lands, Corrigan et al. (2018) overlaid amphibian, bird, and mammal ranges in PAs governed by Indigenous peoples and other governance arrangements in Australia, Brazil, and Namibia, finding similar numbers of nationally recorded species in both types of PA governance. Similarly, in Australia, Brazil, and Canada, amphibian, bird, mammal, and reptile species richness in Indigenous-managed lands were higher than in PAs and randomly selected non-protected areas (Schuster et al., 2019).

At the global level, Garnett et al. (2018) published a dataset of terrestrial lands managed or owned by Indigenous peoples across 87 countries compiled from 127 data sources that has allowed for global evaluations of their importance to conservation outcomes. Indigenous peoples' lands are estimated to cover ≤28% of global land area, overlapping with ≤40% of PAs and accounting for 37% of all remaining lands free of industrial-level human impacts (Garnett et al., 2018). This dataset has been used to assess the importance of Indigenous peoples' lands for 4460 terrestrial mammals' Area of Habitat (O'Bryan et al., 2020), and for 521 primates (Estrada et al., 2022). Indigenous peoples' lands were found to cover ≥10% of the ranges of 2695 mammalian species and >50% of the ranges of 1009 mammals (O'Bryan et al., 2020). They also covered 30% of global primate ranges, compared to 23% for PAs, and overlapped with the ranges of 362 primate species (Estrada et al., 2022).

1.4. Importance of tropical forests

Tropical forests and their associated biodiversity are vital for planetary functioning and human well-being, accounting for at least one-third of terrestrial productivity (Malhi, 2012) and 55% of global terrestrial carbon stocks (Pan et al., 2011). Tropical forests are also home to an estimated half of all global terrestrial biodiversity (Gibson et al., 2011; Pimm & Raven, 2000) and form an integral aspect of the biocultural identities and livelihoods of millions of Indigenous forest peoples (United Nations et al., 2021). They thus have a disproportionately important role in the current biodiversity and climate crises, both in contributing and in mitigating.

Despite their global importance, tropical forests continue to be lost through conversion to intensive industrial agricultural land-uses for commodity export markets (Curtis et al., 2018; Henders et al., 2015; Hoang & Kanemoto, 2021) and industrial-scale resource extraction and mining and the development of large-scale infrastructure (Bebbington et al., 2020; Finer et al., 2008; Giljum et al., 2022; Johnson et al., 2020). Agricultural expansion is concentrated in the tropics (Pendrill et al., 2022), as is industrial mining (Luckeneder et al., 2021), while the increased access into tropical forests provided by linear infrastructure and habitat conversion have contributed to defaunation through over-harvesting of wildlife resources beyond subsistence for urban and international markets (Lee et al., 2020; Tregidgo et al., 2017). Much of this exploitation is concentrated on Indigenous peoples' lands (Anongos et al., 2012; Scheidel et al., 2023), whose land tenure are often insecure, resulting in land conflicts and furthering habitat loss and degradation (Pendrill et al., 2022). As such, tropical forests are a major focus for conservation attention as areas of high biodiversity and carbon value potentially being lost.

1.5. Thesis overview

This thesis thus sets out to evaluate whether Indigenous lands contribute to tropical forest conservation outcomes globally. Acknowledging that articulating Indigeneity is complex especially in Africa (Veber et al., 1993) and Asia (Kingsbury, 1998), in this thesis, I follow the definition applied by Garnett et al. (2018), which also aligns with that of ILO 169. I use the Indigenous Peoples' Land dataset (Garnett et al., 2018) and World Database on Protected Areas (UNEP-WCMC & IUCN, 2020) to identify Indigenous lands, Protected Areas, the overlapping areas of Protected Areas in Indigenous lands, and non-protected areas across tropical Americas, Africa, and Asia. Indigenous lands, as used in this thesis, refer to Garnett et al.'s Indigenous Peoples' Lands that are outside of PAs. I define tropical forests using Dinerstein et al. (2017)'s biomes of Tropical and Subtropical Moist Broadleaf, Dry Broadleaf and Coniferous Forests.

The empirical chapters presented in this thesis represent, to my knowledge, the first evaluations of deforestation and degradation rates and forest integrity in Indigenous lands across the tropics and of terrestrial mammal range coverage by Indigenous peoples' lands. They thus add to the evidence regarding the extent to which Indigenous lands achieve conservation outcomes. This is of significance within the Kunming-Montreal post-2020 Global Biodiversity Framework, since Indigenous lands have the potential to be considered under OECMs should Indigenous communities wish to register their lands as such. Although analyses at global scales rarely contain sufficient details on the contexts of specific places where conservation interventions occur, these global studies still have value in providing a broad overview and basis upon which to conduct more localised and grounded studies and in supporting policy decisions.

In Chapters 2 and 3, I use propensity score matching to identify comparable counterfactual controls of Indigenous lands, PAs, overlapping PAs in Indigenous lands, followed by regression to estimate deforestation and degradation rates (Chapter 2: Reduced deforestation and degradation in Indigenous Lands pan-tropically) and forest integrity (Chapter 3: Indigenous lands in protected areas have high forest integrity across the tropics). Given that the Indigenous Peoples' Land dataset does not necessarily indicate Indigenous land tenure, and that areas indicated on the map may not actually be currently managed or governed by Indigenous peoples, an important clarification is that the use of quasi-experimental methods was not to conduct an impact evaluation and causally attribute conservation outcomes to Indigenous governance. Rather, matching reduced location biases while providing an overview of the state of conservation outcomes in these spaces at the pan-tropical scale. It is arguable if impact evaluations of Indigenous lands could be adequately done, given that many Indigenous communities have used, managed, or governed their lands since before the monitoring of conservation outcomes began.

My third empirical chapter concentrates on faunal diversity (Chapter 4: Indigenous Peoples' Lands are critical for safeguarding vertebrate diversity across the tropics), specifically tropical forest-dependent amphibians, birds, mammals, and reptiles. I focused on the distribution of forest-dependent vertebrates' Area of Habitat and their overlap with Indigenous Peoples' Lands, shifting away from the previous chapters' focus on comparisons with PAs after accounting for location biases. I identified the percentage of each species' range overlap with Indigenous lands, PAs, and the overlapping PAs in Indigenous lands. I also calculated three indices of biodiversity value – species richness, extinction vulnerability, and range-size rarity– and compared these values within Indigenous peoples' lands and within the 10 km buffer zone outside.

The last chapter (Chapter 5: General discussion) brings together these three empirical chapters and discusses the implications of my findings for addressing the biodiversity and climate crises generally. I consider Indigenous-led approaches in conservation and the issue of decolonising conservation before concluding the thesis.

1.6. My positionality

Although it is not typical for a natural sciences thesis to contain a section on positionality, given the nature of my research on Indigenous peoples' issues, I felt it was critical for inclusion. Positionality refers to various intersecting dimensions of one's identity (e.g. ethnicity, nationality, sexuality, gender) that affects the opportunities and privileges that one has and contributes to the shaping of one’s worldviews (cosmology), ways of understanding the world and reality (ontology), and understanding the nature of knowledge (epistemology) (Holmes, 2020; Moon & Blackman, 2014). They have bearing on the values that we hold as individuals, shaped by the society that we were brought up in, and on the relationships that we have with others. Our positionality also has implications for the way research is done, as they affect what research questions one may consider worth exploring, the methods or methodologies employed in answering those questions, and the outputs generated from the research (Meinherz et al., 2020; Tuhiwai Smith, 2012).

This section thus answers the call for more conservationists to be reflexive (thinking that allows researchers to recognise the effect that researchers have on their research; Brittain et al., 2020) and consider our positionality in research (Archer et al., 2022; Boyce et al., 2022; Moon et al., 2019). This involves reflecting on the assumptions that we as researchers hold given our training in the dominant system of knowledge production and the normative values we hold as part of our society, and acknowledging the power imbalances between the researcher and the researched. However, since my thesis did not entail fieldwork nor research partners beyond my supervisors, my positionality, as a woman who appears to be of East Asian ethnicity, has had little influence on the relationships that I have had with others over the course of this research.

Nonetheless, I recognise that as a PhD researcher in a British university, a privileged site of knowledge production, I hold certain privileges and power, particularly in the potential to shape discourses and policy. This privilege includes access to data, computing power, and the knowledge and training to use them. Further, having been educated in mainstream sciences and belonging to dominant political economies, my knowledge is typically accepted as truth, assumed to be right, and not challenged. By contrast, Indigenous peoples’ knowledges and worldviews are often invisibilised and not recognised as valuable (Tuhiwai Smith, 2012). As such, over the course of the PhD, I have tried to read, learn, and understand Indigenous peoples’ perspectives on conservation. My thesis thus attempts to highlight efforts by Indigenous peoples and support their efforts in gaining greater public recognition for their contributions and rights to their autonomy, self-determinations, and lands, as a way of using my privilege without harming communities less privileged than I.

Chapter 2

Reduced deforestation and degradation in Indigenous Lands pan-tropically

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2.1. 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.

2.2. Introduction

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 (Weisse & Goldman, 2020). Tropical forests are central in preserving global biodiversity and retain ~55% of global terrestrial carbon stocks (Gibson et al., 2011; Pan et al., 2011). They are also essential to the biocultural identities and livelihoods of Indigenous forest peoples ((United Nations et al., 2021). Alongside increasing amounts of forest lost through commodity frontier expansions for agribusiness, particularly in Southeast Asia, Central and South America, and West Africa (Curtis et al., 2018), there is a lack of legal recognition of Indigenous rights and respect for customary tenure in many countries (United Nations et al., 2021). This has resulted in land grabs, violence and killings of Indigenous and local peoples defending their land and forests (Larsen et al., 2020).

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 (Tauli-Corpuz et al., 2020). 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) (Dinerstein et al., 2019; Tauli-Corpuz et al., 2020).

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 (CBD, 2020; Dudley et al., 2018). Although there are concerns that expanding area-based targets will further the injustices experienced by Indigenous Peoples and local communities (Rainforest Foundation UK et al., 2021; Tauli-Corpuz et al., 2020), it is also a transformative opportunity (Reyes-García et al., 2021). 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 (ICCA Consortium, 2021; Reyes-García et al., 2021).

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 (Garnett et al., 2018) and at least 36% of Intact Forest Landscapes (Fa et al., 2020). In Latin America, ILs have reduced deforestation and degradation rates (Blackman & Veit, 2018; Vergara-Asenjo & Potvin, 2014; Walker et al., 2020), often more so than in state-managed protected areas (PAs) (Jusys, 2018; Nolte et al., 2013; Schleicher et al., 2017). 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 (Blackman & Veit, 2018). 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 (Jackson & Warren, 2005; United Nations, 2009). 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 (United Nations et al., 2021).

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 (Garnett et al., 2018). 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 (Vancutsem et al., 2021), 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 (Hansen et al., 2013) 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 (Stuart & Rubin, 2008), whilst accounting for confounding factors including location and accessibility, that affect deforestation and PA establishment and can overestimate intervention effectiveness (Nolte et al., 2013; Pfaff et al., 2014).

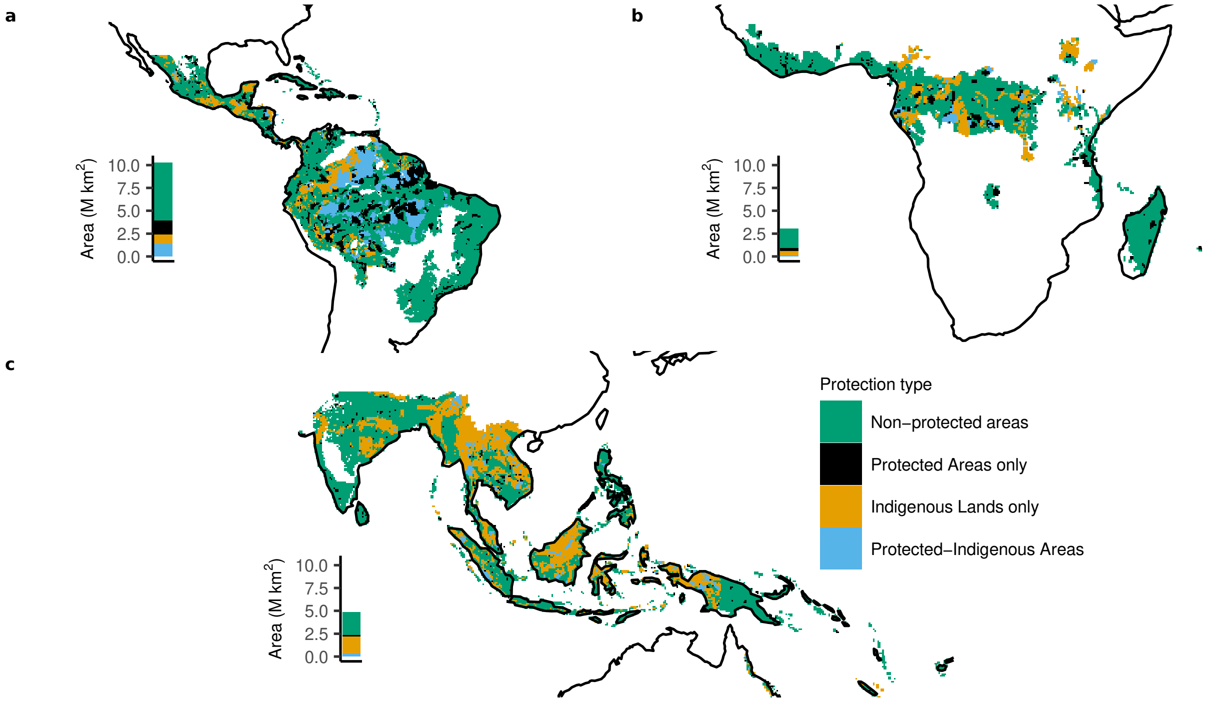


Figure 1. Indicative map of the different protection types across tropical moist forests within our analysis, coarsened to 30 km resolution where each pixel represents the dominant type. The resolution is intentionally coarsened so that boundaries are imprecise, as boundaries of Indigenous territories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

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).

2.3. 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.

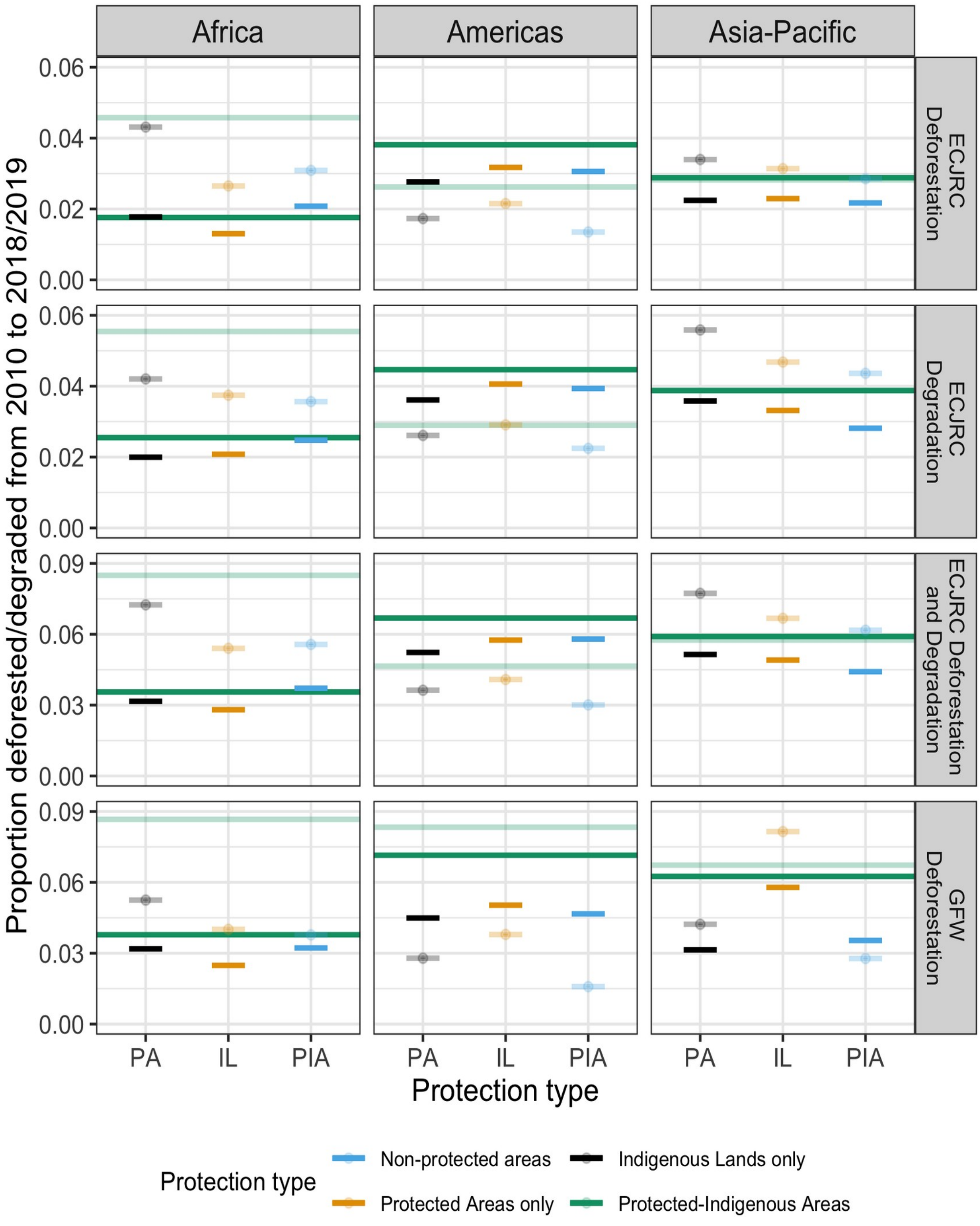


Figure 2. Mean estimated deforestation rates from 2010 to 2019 (or 2018 for GFW data) predicted from GAMM regional models of protection types for GFW deforestation rates, combined ECJRC deforestation and degradation rates, and separate ECJRC deforestation and degradation rates across tropical moist forest 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.

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).

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 (Leberger et al., 2020). 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 (Borrini-Feyerabend et al., 2013). 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. (2018) 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 (Dubertret & Alden Wily, 2015; RRI, 2015). 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.

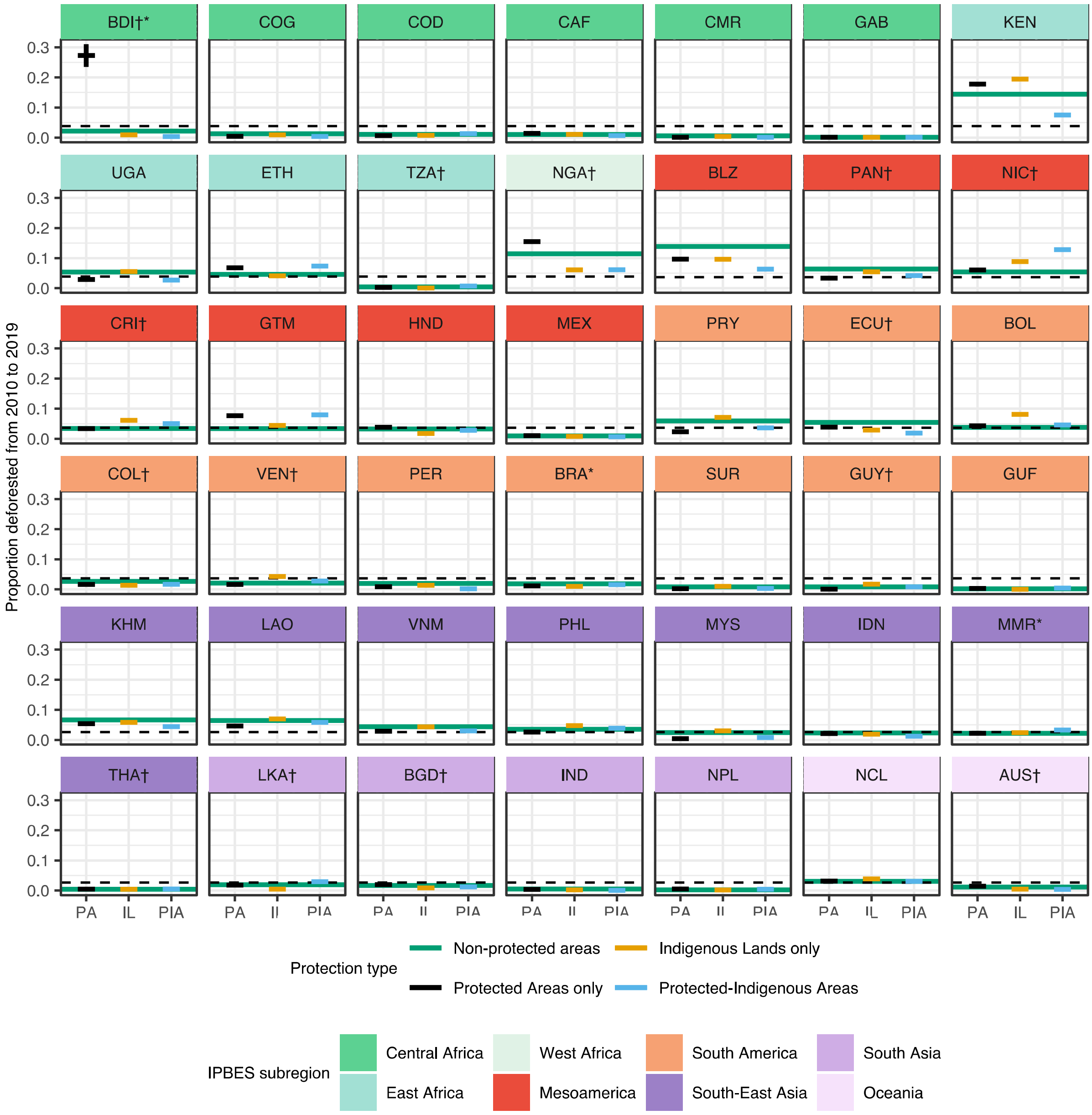


*Figure 3. ECJRC deforestation rates predicted from GAMM models of protection types for each country. Dashed line refers to regional mean deforestation rate of non-protected areas, vertical line shows standard error. Values below the green line represent avoided deforestation relative to non-protected 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.

2.4. 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 (Tauli-Corpuz et al., 2020) . 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 (ICCA Consortium, 2021; Reyes-García et al., 2021), and suggests the dovetailing of mainstream conservation efforts and Indigenous protection is a plausible ambition.

2.4.1. Deforestation and degradation on Indigenous Lands

Despite ongoing encroachment on Indigenous Lands (AIPP et al., 2020), 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 (Blackman & Veit, 2018; Vergara-Asenjo & Potvin, 2014), 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 (Domínguez & Luoma, 2020). 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 (Pyhälä et al., 2016) (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 (Pearson et al., 2017) and doubles biodiversity losses from deforestation (Barlow et al., 2016). 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 (Hansen et al., 2020). 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 (Benítez-López et al., 2019; Milodowski et al., 2021). The complexities of monitoring forest degradation using remote sensing not withstanding (Miettinen et al., 2014), 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 (Vancutsem et al., 2021). 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 (Yuliani et al., 2018).

In many Indigenous communities across the world, there exists both formal and informal institutions for governing forest commons and resources (Berkes, 2017) and monitoring forest access (Sheil et al., 2015). 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 (Sasaoka & Laumonier, 2012). Similarly, in Ghana, the traditional beliefs and practices of the Ashantis prohibit overexploitation of their forests (Asante et al., 2017), while the Indigenous peoples in the Xingu, Brazil, mobilise to keep out intruders (Schwartzman et al., 2013). 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.

2.4.2. 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 (United Nations et al., 2021). 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 (United Nations et al., 2021). Titled Miskitu communities in Bosawas Biosphere Reserve, Nicaragua, were better able to control agricultural expansion threat from mestizo colonisers than were untitled Miskitu communities (Hayes & Murtinho, 2008). However, increasing encroachment and land grabs linked to illicit drug trafficking and the lack of state support have increased deforestation pressure on Indigenous territories (Bryan, 2019; Tellman et al., 2020), 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 (Robinson et al., 2018; Seymour et al., 2014; Tseng et al., 2021). 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 (Smith et al., 2017). The process of formalizing land tenure may also bring additional problems (Larson & Springer, 2016), such as reducing community autonomy and security (Arizona et al., 2019).

2.4.3. 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 (Malavasi, 2020). 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 (Witter & Satterfield, 2019). 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 (Dutta et al., 2021). 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 (Herrera et al., 2019). 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 (Hansen et al., 2020). 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 (Benítez-López et al., 2019).

2.4.4. Conclusion

Protected areas and Indigenous territories are often threatened by the same macroeconomic political forces (Bebbington et al., 2018). Planned investment increases in agriculture, economic growth-inducing infrastructure, and road networks will likely accelerate deforestation and biodiversity loss in the coming decades (Johnson et al., 2020), further increasing pressures both inside and outside protected zones. For instance, high deforestation rates in Cambodia linked to large-scale land acquisitions (Davis et al., 2015) 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 (Conigliani et al., 2018) 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.

2.5. 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.

2.5.1. Data

Deforestation and degradation from 2010 to 2019 Tropical Moist Forest Subtypes were downloaded from the European Commission Joint Research Centre (Vancutsem et al., 2021). 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 (GLAD, 2013) and 2011 to 2018 forest loss data (v1.6) (Global Forest Watch, 2018) were downloaded from Global Forest Watch (Hansen et al., 2013). We defined pixels as being forested in 2010 using 25% tree canopy cover threshold, as recommended by (Hansen et al., 2010) 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 on Protected Areas Jan 2020 version was obtained from UNEP-WCMC & IUCN (2020) using the R package `wdpar` (Hanson, 2020) 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. (2018). We acknowledge that Indigenous lands were mapped based on publicly available information (Garnett et al., 2018) 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 (Global Forest Watch, 2019; Transparent World & Global Forest Watch, 2019) from our analysis.

Matching covariates

Following previous studies (e.g. Joppa & Pfaff, 2009; Nelson & Chomitz, 2011; Nolte et al., 2013; Schleicher et al., 2017), 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 (GLAD, 2013) 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 (Global Forest Watch, 2015) 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. (2018).These influence likelihood of deforestation and PA locations (Joppa & Pfaff, 2009).

Travel time to nearest populated area with more than 5000 population in 2015 was obtained from Nelson et al. (2019). Access to markets and transport hubs influence land-use change decisions.

Distance to roads was calculated from CIESIN (2013) gROADS dataset in ArcMap using Euclidean distance, as proximity to roads is a major driver of deforestation.

Population density in 2010 from WorldPop (Lloyd et al., 2017) was included to control for forest pressure from local human populations.

Countries from GADM (Global Administrative Areas, 2018) 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.

2.5.2. 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 (Stuart & Rubin, 2008).

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 (Ho et al., 2018) 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 (Wood, 2019), 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.

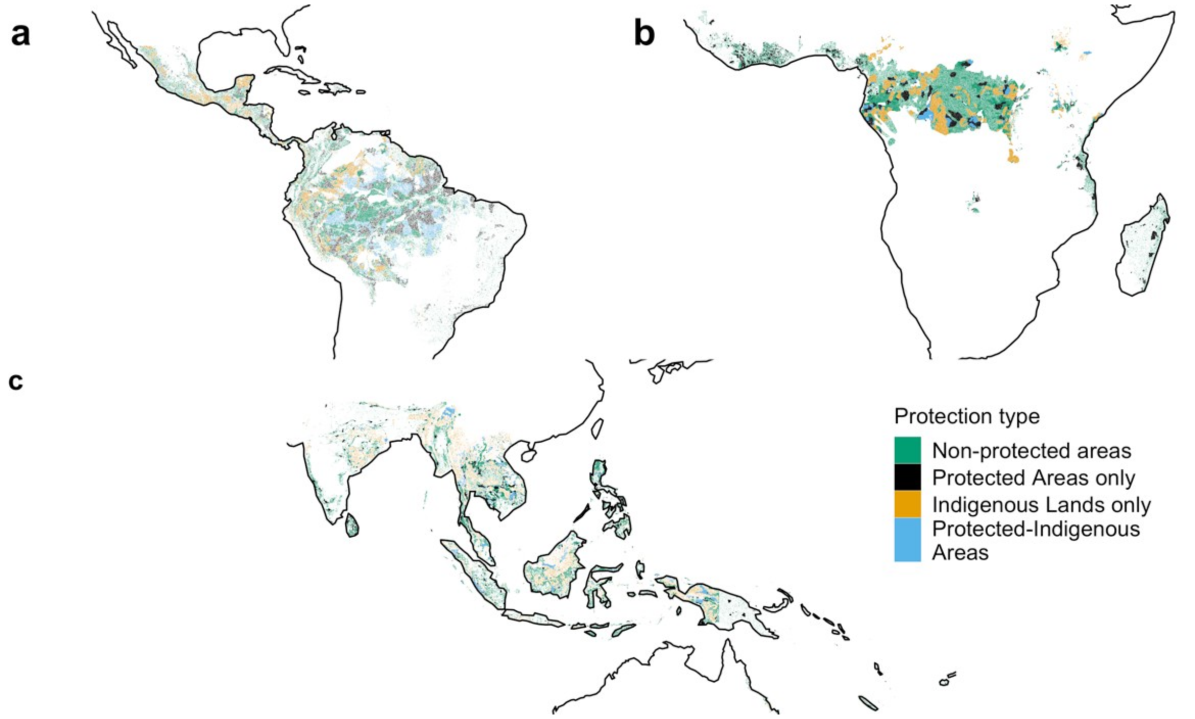
2.6. Data availability

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).

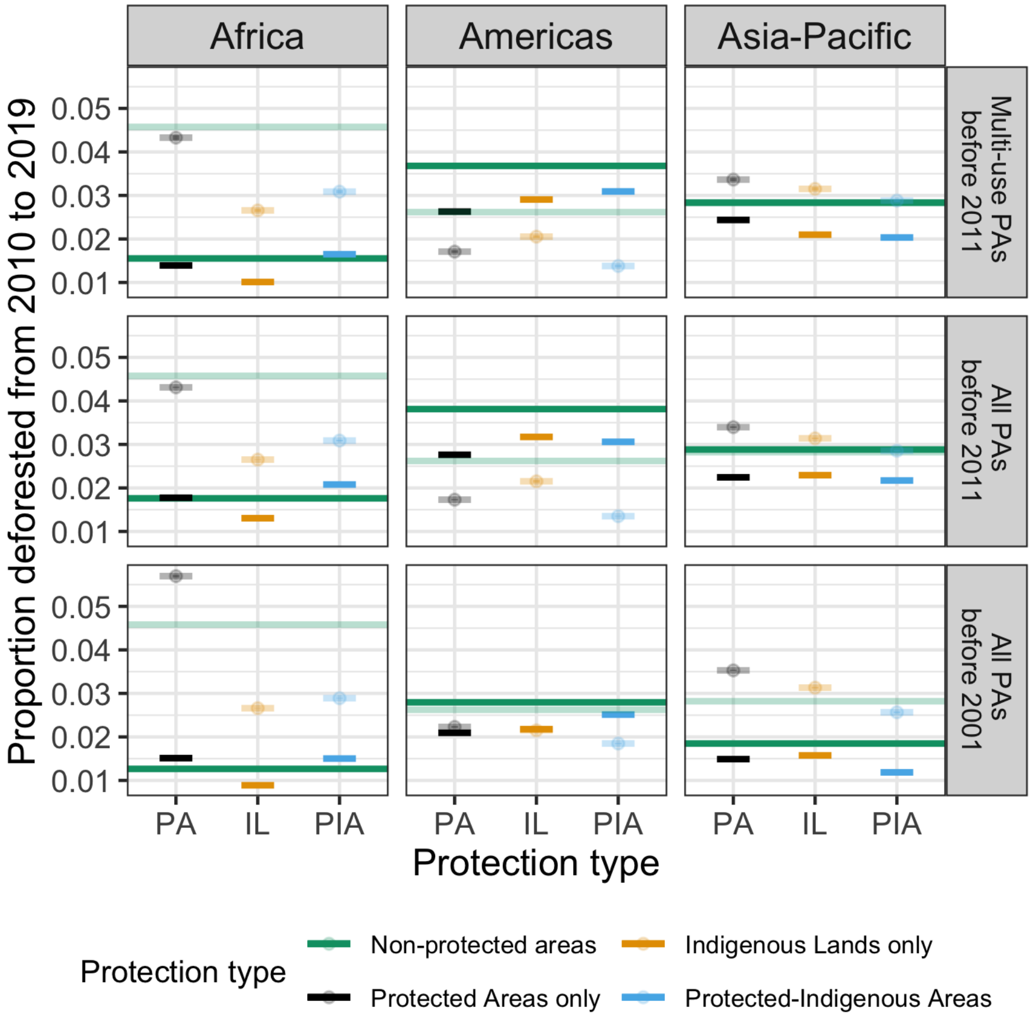
2.7. Code availability

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

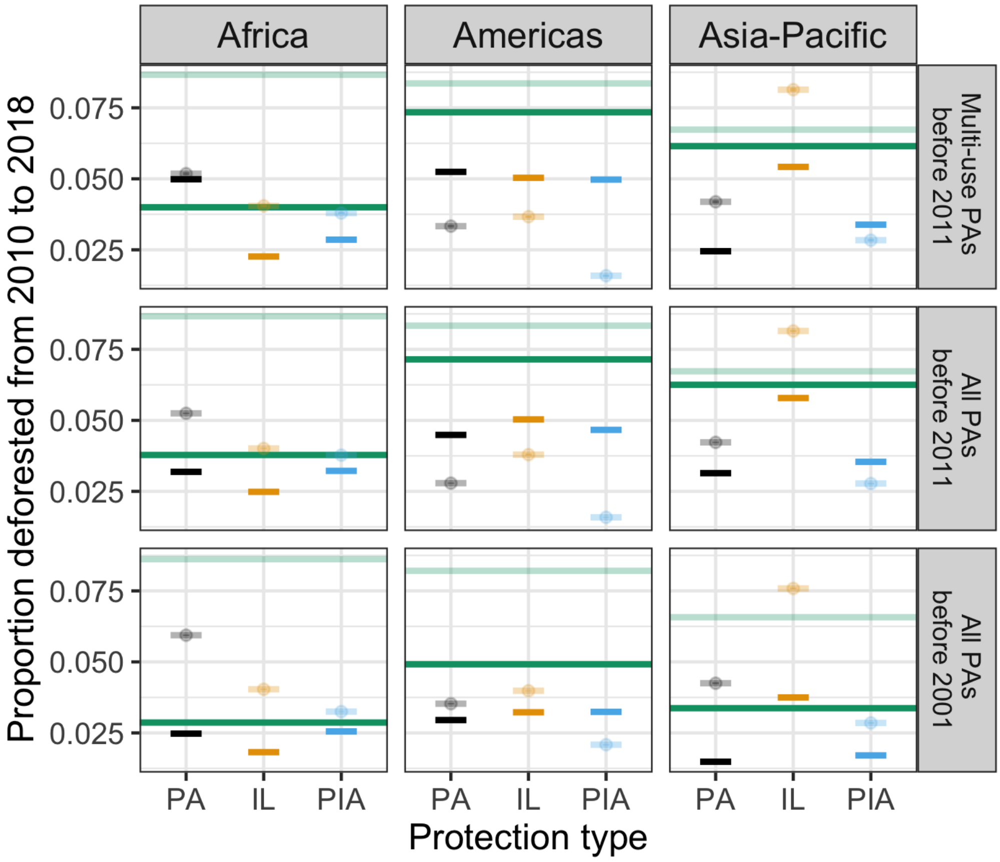
2.8. Supplementary Information



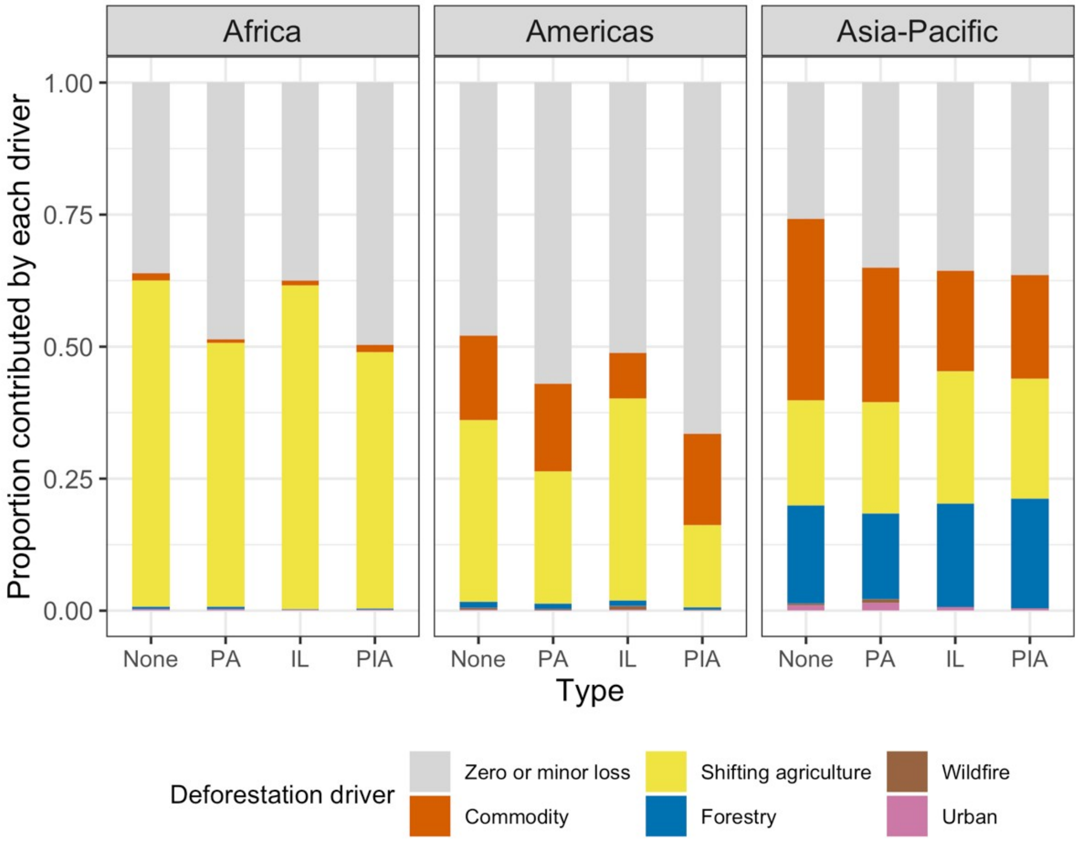
Supplementary Figure 1. Map of the matched data of different protection types at tropical moist forest extents, at coarsened resolutions to obscure boundaries. a, the Americas. b, Africa. c, Asia-Pacific



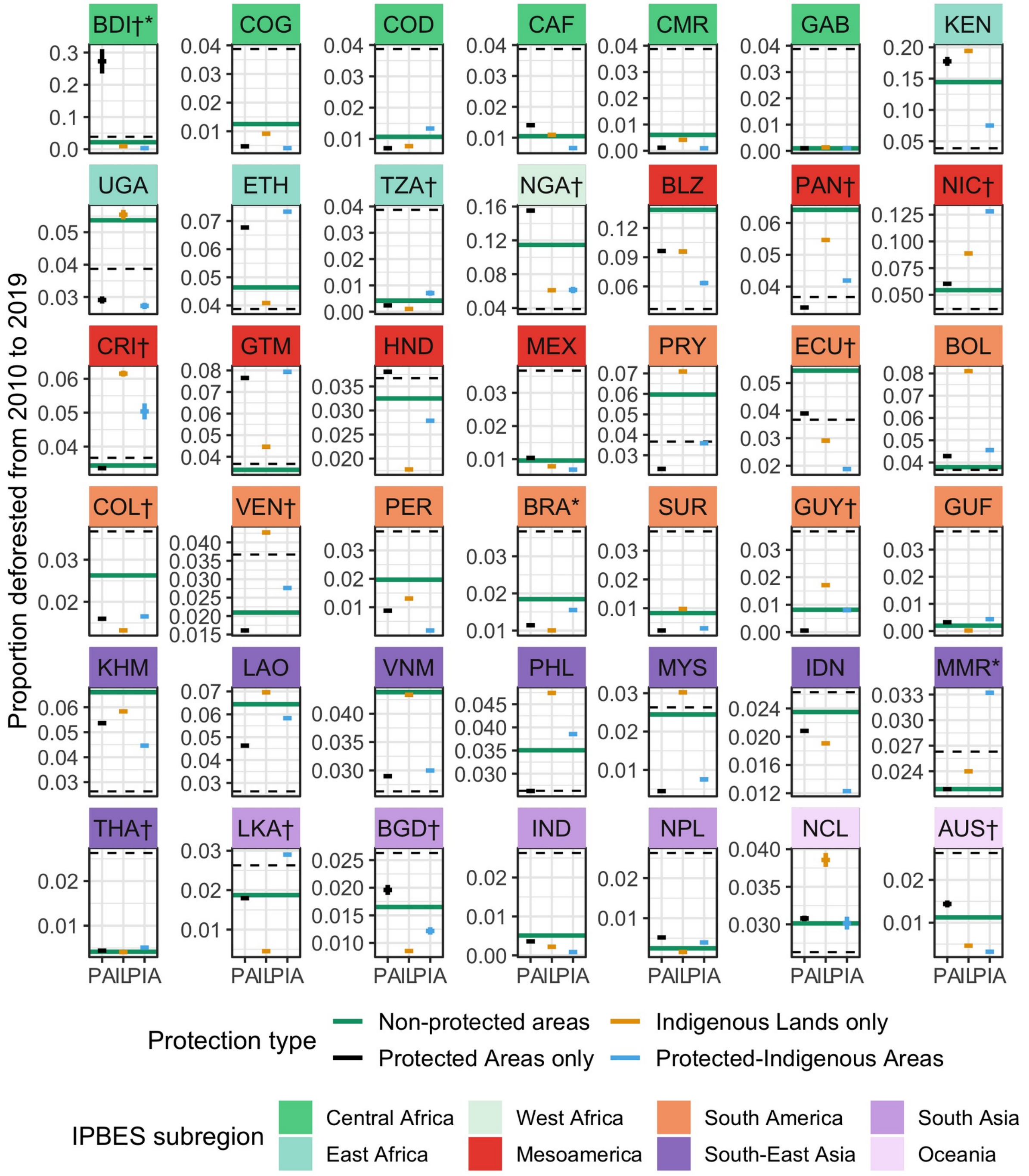
Supplementary Figure 2. Mean estimated deforestation rates from 2010 to 2019 predicted from GAMM regional models of protection types for ***ECJRC deforestation rates*** *before matching (in translucent) and after regression (in solid) across tropical moist forest extents for multi-use PAs, all PAs established before 2011, and all PAs established before 2001.*



Supplementary Figure 3. Mean estimated deforestation rates from 2010 to 2018 predicted from GAMM regional models of protection types for ***GFW deforestation rates*** *before matching (in translucent) and after regression (in solid) across tropical moist forest extents for multi-use PAs, all PAs established before 2011, and all PAs established before 2001. Legend as with Supplementary Figure 2.*

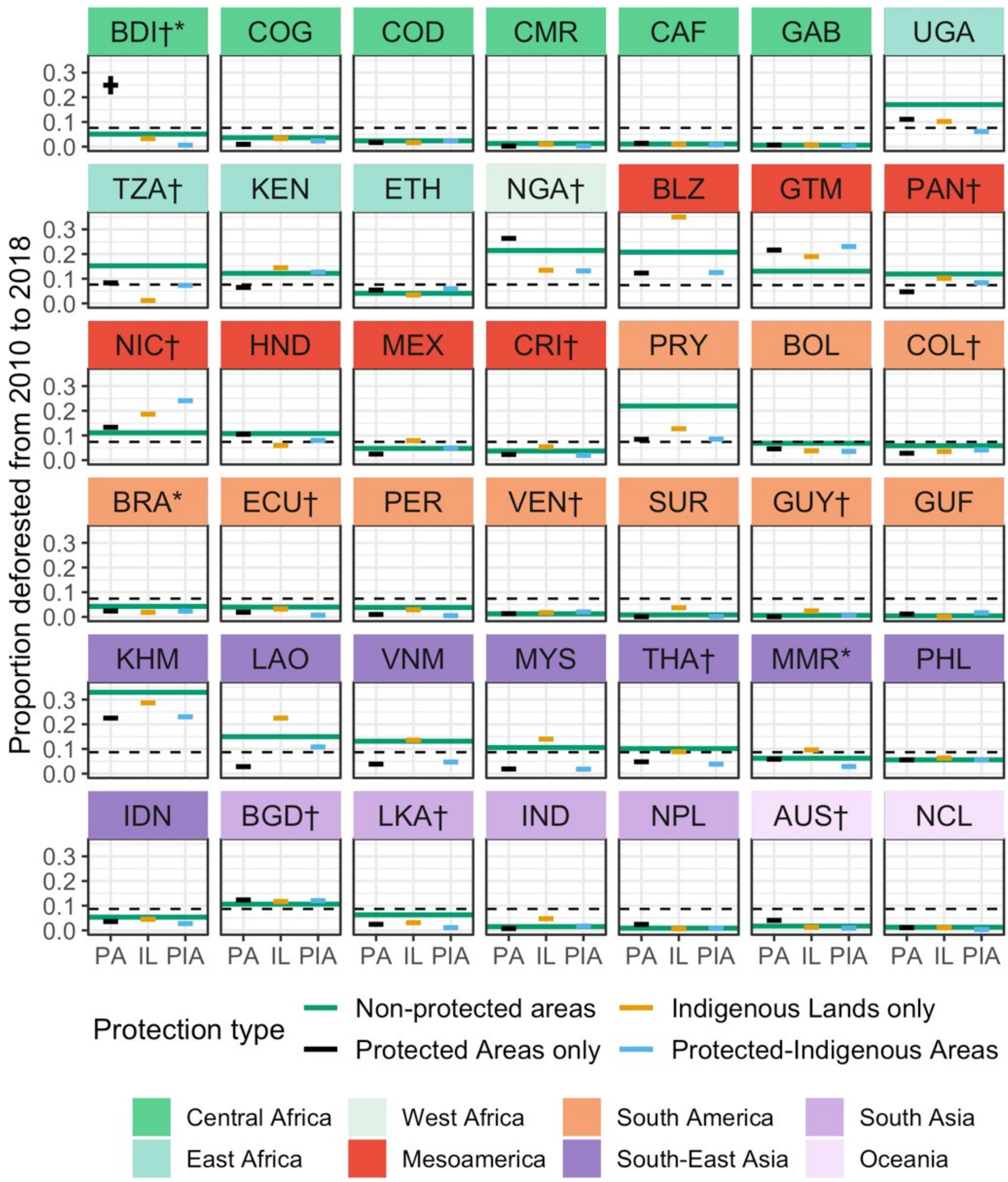


Supplementary Figure 4. Classified drivers of forest loss from Curtis et al. 2018 within each protection type for each region in our study of the matched data at tropical moist forest extents, for PAs that were established before 2011.



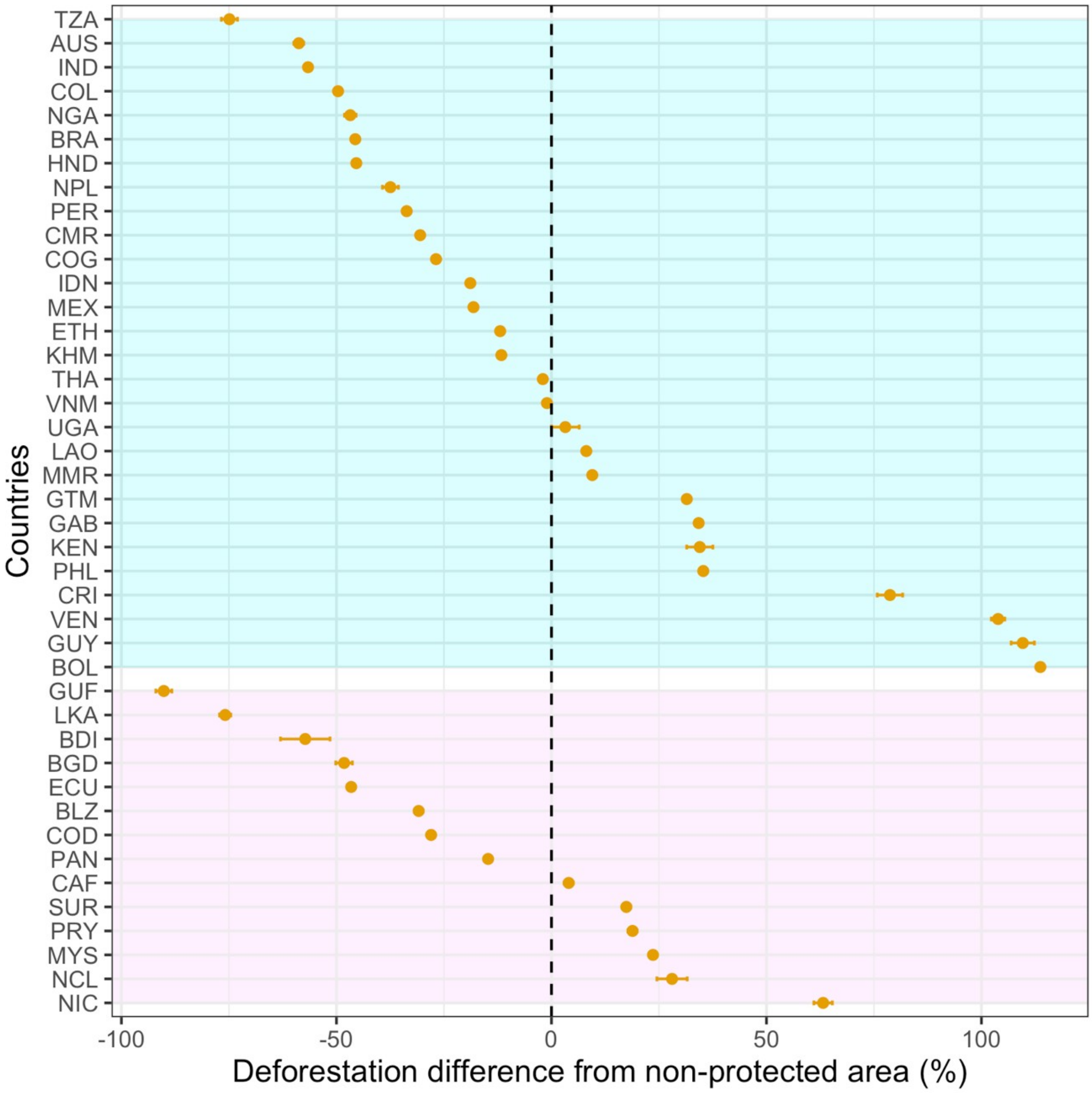
*Supplementary Figure 5. **ECJRC deforestation rates** predicted from GAMM models of protection types for each country. Dashed line refers to regional mean deforestation rate of non-protected areas, vertical line shows standard error, **note different y-axis scales**. Values below the green line represent 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.

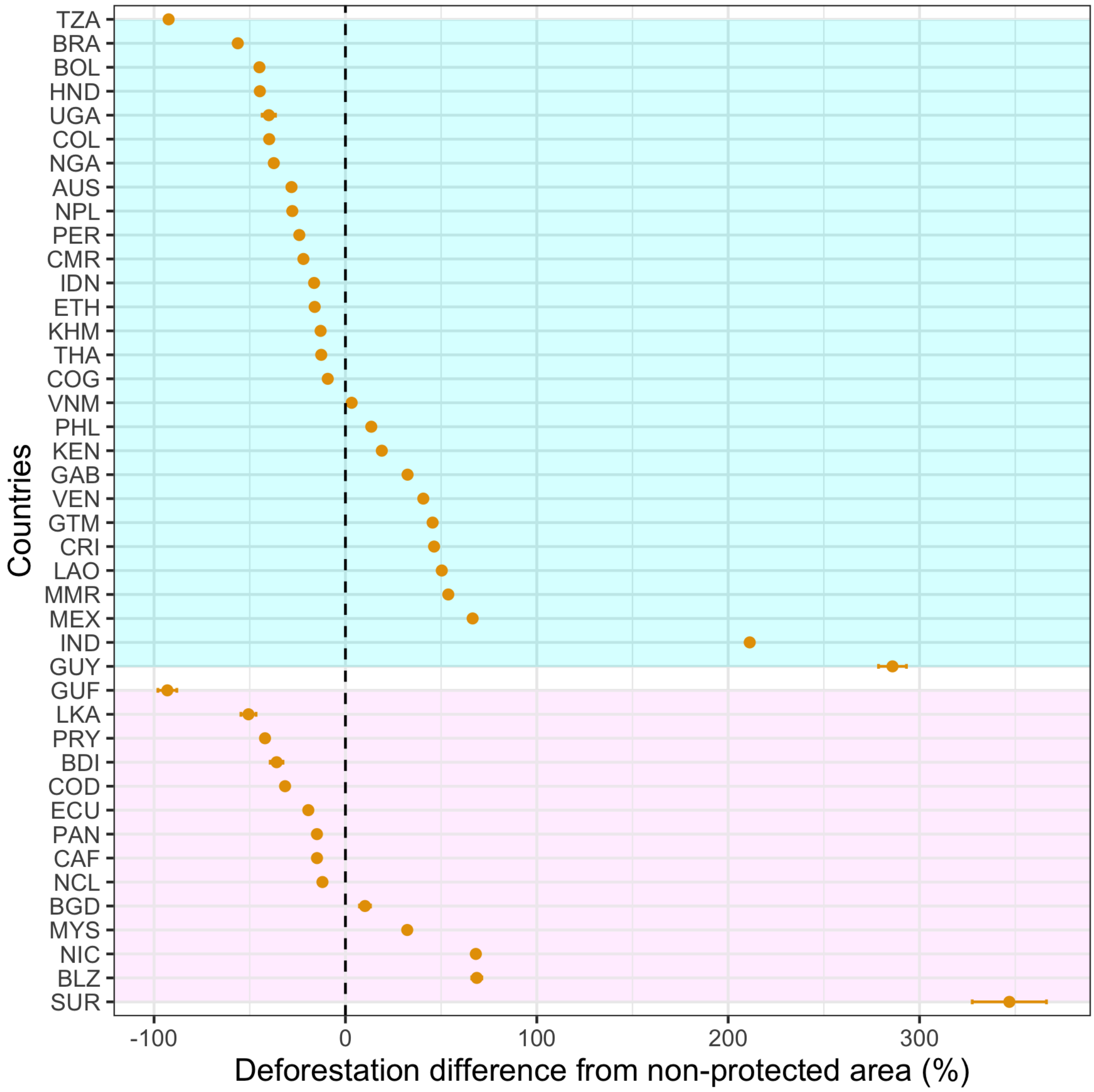


Supplementary Figure 6. ***GFW deforestation rates*** *predicted from GAMM models of protection types for each country. Dashed line refers to regional mean deforestation rate of non-protected areas, vertical line shows standard error. Values below the green line represent avoided deforestation relative 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; CMR Cameroon; CAF Central African Republic; GAB Gabon; UGA Uganda; TZA Tanzania; KEN Kenya; ETH Ethiopia; NGA Nigeria; BLZ Belize; GTM Guatemala; PAN Panama; NIC Nicaragua; HND Honduras; MEX Mexico; CRI Costa Rica; PRY Paraguay; BOL Bolivia;*

COL Colombia; BRA Brazil; ECU Ecuador; PER Peru; VEN Venezuela; SUR Suriname; GUY Guyana; GUF French Guiana; KHM Cambodia; LAO Laos; VNM Vietnam; MYS Malaysia; THA Thailand; MMR Myanmar; PHL Philippines; IDN Indonesia; BGD Bangladesh; LKA Sri Lanka; IND India; NPL Nepal; AUS Australia; NCL New Caledonia.



Supplementary Figure 7. Difference in ***ECJRC deforestation rates*** *(error bars represent percent standard error) in Indigenous lands relative to non-protected areas for countries where state governments have recognised Indigenous land ownership/management (in blue shades) and where they have not (in pink shades), according to LandMark and RRI data. A positive value means that Indigenous lands have higher deforestation rates than non-protected areas.*



*Supplementary Figure 8. Difference in **GFW deforestation rates** (error bars represent percent standard error) in Indigenous lands relative to non-protected areas for countries where state governments have recognised Indigenous land ownership/management (in blue shades) and where they have not (in pink shades), according to LandMark and RRI data. A positive value means that Indigenous lands have lower deforestation rates than non-protected areas.*

*Supplementary Table 1. Area of different protection types (square kilometres) for countries included in analysis at tropical moist forest extents, where forest is considered at 25% tree canopy cover threshold at the native resolution using GFW data. * refer to countries included in the country analysis, where matching and regression were done at the country level*

ISO3	Country	IPBES Subregion	IPBES Region	Non-protected area	PA	IL	PIA
		AFRICA		**2162901.42**	**300965.01**	**475739.15**	**103929.04**
BDI*	Burundi	Central Africa	Africa	9342.47	89.35	538.04	252.19
CAF*	Central African Republic	Central Africa	Africa	31951.78	8755.35	12738.5	7342.64
CIV	Côte d'Ivoire	West Africa	Africa	102640	33178.95	NA	NA
CMR*	Cameroon	Central Africa	Africa	161351.71	12570.36	49784.99	4332.82
COD*	Democratic Republic of the Congo	Central Africa	Africa	700117.48	99010.83	243029.7	59825.96
COG*	Republic of Congo	Central Africa	Africa	141920.44	17404.33	40985.51	2186.06
COM	Comoros	East Africa and adjacent islands	Africa	1381.41	3.36	NA	NA
ETH*	Ethiopia	East Africa and adjacent islands	Africa	8677.63	2581.63	44516.49	3930.64
GAB*	Gabon	Central Africa	Africa	94658.91	31042.15	60469.49	15858.02
GHA	Ghana	West Africa	Africa	57510.08	10694.04	NA	NA
GIN	Guinea	West Africa	Africa	39252.84	4868.3	NA	NA
GNQ	Equatorial Guinea	Central Africa	Africa	20779.39	3879.82	NA	NA
KEN*	Kenya	East Africa and	Africa	31160.22	3462.63	14538.08	6295.07

		adjacent islands					
LBR	Liberia	West Africa	Africa	89534.72	2816.06	NA	NA
MDG	Madagascar	East Africa and adjacent islands	Africa	396695.91	22457.34	NA	NA
MOZ	Mozambique	Southern Africa	Africa	38472.73	5126.28	NA	NA
MUS	Mauritius	East Africa and adjacent islands	Africa	1300.25	23.39	NA	NA
NGA*	Nigeria	West Africa	Africa	91963.24	14203.98	6972.51	1662.52
REU	Reunion	East Africa and adjacent islands	Africa	564.68	1311.82	NA	NA
RWA	Rwanda	East Africa and adjacent islands	Africa	8179.27	708.46	1060.82	192.59
SLE	Sierra Leone	West Africa	Africa	42907.78	1245.51	NA	NA
SSD	South Sudan	North Africa	Africa	2549.44	1281.13	NA	NA
TGO	Togo	West Africa	Africa	4569.18	554.6	3.4	NA
TZA*	Tanzania	East Africa and adjacent islands	Africa	55190.83	15612.58	265.48	379.13
UGA*	Uganda	East Africa and adjacent islands	Africa	19082.19	3405.08	836.14	1671.41
ZMB	Zambia	Southern Africa	Africa	11146.84	4677.71	NA	NA
		AMERICAS		**6345237.53**	**1538290.2**	**1030640.18**	**1350718.35**
ARG	Argentina	South America	Americas	67.56	6.15	92.83	0.78
ATG	Antigua and Barbuda	Caribbean	Americas	30.35	3.29	NA	NA

BHS	Bahamas	Caribbean	Americas	3128.67	420.7	NA	NA
BLZ*	Belize	Mesoamerica	Americas	4442.49	3889.72	3679.52	1141.34
BOL*	Bolivia	South America	Americas	299000.95	92694.54	80400.35	40527.1
BRA*	Brazil	South America	Americas	3668259.35	955014.29	22150.05	937922.06
COL*	Colombia	South America	Americas	499890.69	85655.66	236724.79	36357
CRI*	Costa Rica	Mesoamerica	Americas	30073.79	6686.06	3371.53	1152.94
CUB	Cuba	Caribbean	Americas	77840.18	6757.69	NA	NA
DMA	Dominica	Caribbean	Americas	258.31	72.99	2.49	NA
DOM	Dominican Republic	Caribbean	Americas	31652.52	8143.35	NA	NA
ECU*	Ecuador	South America	Americas	115631.1	15218.13	47433.93	8713.32
GRD	Grenada	Caribbean	Americas	176.54	1.68	NA	NA
GTM*	Guatemala	Mesoamerica	Americas	25209.46	11462.73	52780.17	5672.01
GUF*	French Guiana	South America	Americas	36928.81	34234.85	288.96	6325.26
GUY*	Guyana	South America	Americas	153053.08	3831.11	16309.2	6161.54
HND*	Honduras	Mesoamerica	Americas	55326.64	11548	15573.7	7839.02
HTI	Haiti	Caribbean	Americas	22062.71	366.06	NA	NA
JAM	Jamaica	Caribbean	Americas	7768.91	648.06	NA	NA
LCA	Saint Lucia	Caribbean	Americas	307.7	46.63	NA	NA
MEX*	Mexico	Mesoamerica	Americas	503782.79	65397.92	195491.6	21452.55
MTQ	Martinique	Caribbean	Americas	229.32	256.34	NA	NA
NIC*	Nicaragua	Mesoamerica	Americas	48179.36	22838.32	10216.22	12788.92

PAN*	Panama	Mesoamerica	Americas	33019.7	3474.92	18353.21	8341.07
PER*	Peru	South America	Americas	436261.34	127821.83	172621.48	18089.73
PRI	Puerto Rico	Caribbean	Americas	7377.73	127.34	NA	NA
PRY*	Paraguay	South America	Americas	75516.45	571.48	3063.27	1357.29
SLV	El Salvador	Mesoamerica	Americas	12560.98	683.83	4139.01	3.34
SUR*	Suriname	South America	Americas	69659.76	10019.63	45082.36	6212.68
TTO	Trinidad and Tobago	Caribbean	Americas	2641.79	874.41	NA	NA
VCT	Saint Vincent and the Grenadines	Caribbean	Americas	92.16	18.43	NA	NA
VEN*	Venezuela	South America	Americas	124761.27	69496.69	102865.5	230660.39
VIR	Virgin Islands, U.S.	Caribbean	Americas	45.08	7.35	NA	NA
		ASIA-PACIFIC		**2525594.82**	**170084.22**	**1878049.83**	**295003.52**
AUS*	Australia	Oceania	AsiaPacific	10754.09	975.37	3275.09	841.7
BGD*	Bangladesh	South Asia	AsiaPacific	86785.74	408.19	11639.27	612.94
BRN	Brunei	South-East Asia	AsiaPacific	3913.53	802.42	3.5	0.12
BTN	Bhutan	South Asia	AsiaPacific	1963.62	884.4	NA	NA
CHN	China	North-East Asia	AsiaPacific	144.35	NA	661.79	5.61
FJI	Fiji	Oceania	AsiaPacific	11344.49	203.13	NA	9.86
IDN*	Indonesia	South-East Asia	AsiaPacific	617781.66	34873.12	632180.59	132871.44
IND*	India	South Asia	AsiaPacific	523281.9	21920.85	239871.49	13171.54

KHM*	Cambodia	South-East Asia	AsiaPacific	63368.19	21048.57	36522.22	18264.49
LAO*	Laos	South-East Asia	AsiaPacific	38414.74	3644.29	141756.02	30480.77
LKA*	Sri Lanka	South Asia	AsiaPacific	37231.51	9166.3	115.13	236.35
MMR*	Myanmar	South-East Asia	AsiaPacific	188333.5	4479.03	327085.63	22915.88
MYS*	Malaysia	South-East Asia	AsiaPacific	48604.96	3361.51	137940.8	12178.31
NCL*	New Caledonia	Oceania	AsiaPacific	6594.25	1494.13	3622.35	304.63
NPL*	Nepal	South Asia	AsiaPacific	6692.27	1072.5	14311.44	110.54
PHL*	Philippines	South-East Asia	AsiaPacific	148048.67	26730.13	19173.4	5683.15
PNG	Papua New Guinea	Oceania	AsiaPacific	412681.11	7237.5	NA	NA
SGP	Singapore	South-East Asia	AsiaPacific	434.27	10.32	NA	NA
SLB	Solomon Islands	Oceania	AsiaPacific	22934.97	133.94	NA	NA
THA*	Thailand	South-East Asia	AsiaPacific	160696.16	25517.72	182308.51	48440.62
TLS	Timor-Leste	South-East Asia	AsiaPacific	11411.52	1384.61	NA	NA
VNM*	Vietnam	South-East Asia	AsiaPacific	114756.32	4443.65	127582.6	8800.56
VUT	Vanuatu	Oceania	AsiaPacific	9423.01	292.56	NA	NA

*Supplementary Table 2. Mean estimated deforestation rates from 2010 to 2019 (or 2018 for GFW data) for the five samples of regional analyses, standard error of the mean in brackets, and percent difference and percent standard error from non-protected area in square brackets. *reduction in number of PA pixels in Africa and Asia-Pacific meant that we took a sample of the data only for Americas and Asia-Pacific, and the standard errors presented here are from the GAMM models rather than standard error of the mean of 5 samples.*

		Africa				**Americas**				**Asia-Pacific**			
Dataset used	**PA inclusion**	**Non**	**PA**	**IL**	**PIA**	**Non**	**PA**	**IL**	**PIA**	**Non**	**PA**	**IL**	**PIA**
ECJRC deforestation	PAs established before 2011	0.0176 (0.0002)	0.0177 (0.0001) [0.8±1.5 %]	0.0130 (0.0001) [-25.9±1.1%]	0.0208 (0.0002) [18.1±2%]	0.0381 (0.0004)	0.0276 (0.0004) [-27.5±1.2 %]	0.0317 (0.0004) [-16.8±1.3 %]	0.0306 (0.0004) [-19.7±1.4 %]	0.0288 (0.0002)	0.0224 (0.0002) [-22.2±0.8 %]	0.0229 (0.0002) [-20.5±0.9 %]	0.0217 (0.0002) [-24.7±0.8 %]
ECJRC degradation		0.0255 (0.0004)	0.0200 (0.0003) [-21.7±1.8 %]	0.0208 (0.0002) [-18.4±1.5%]	0.0248 (0.0004) [-2.8±2.1%]	0.0447 (0.0005)	0.0361 (0.0004) [-19.1±1.3 %]	0.0406 (0.0005) [-9.1±1.4%]	0.0393 (0.0002) [-11.9±1%]	0.0388 (0.0002)	0.0358 (0.0002) [-7.6±0.8%]	0.0332 (0.0002) [-14.5±0.7 %]	0.0282 (0.0002) [-27.4±0.7 %]
ECJRC deforestation and degradation		0.0356 (0.0005)	0.0316 (0.0004) [-11.1±1.7 %]	0.0280 (0.0002) [-21.2±1.3%]	0.0371 (0.0004) [4.4±2%]	0.0668 (0.0007)	0.0523 (0.0006) [-21.8±1.3 %]	0.0575 (0.0007) [-13.9±1.4 %]	0.0580 (0.0003) [-13.3±1.1 %]	0.0591 (0.0002)	0.0514 (0.0003) [-12.9±0.6 %]	0.0490 (0.0004) [-16.9±0.7 %]	0.0442 (0.0003) [-25.2±0.6 %]
GFW deforestation		0.0378 (0.0006)	0.0319 (0.0007) [-15.7±2.2 %]	0.0248 (0.0002) [-34.3±1.1%]	0.0322 (0.0003) [-14.8±1.5 %]	0.0715 (0.0013)	0.0449 (0.0008) [-37.2±1.6 %]	0.0503 (0.0006) [-29.6±1.5 %]	0.0467 (0.0008) [-34.7±1.6 %]	0.0625 (0.0011)	0.0314 (0.0004) [-49.8±1.1 %]	0.0579 (0.0008) [-7.5±2%]	0.0354 (0.0003) [-43.4±1.1 %]

ECJRC deforestation		0.0155 (0.00005)	0.0139 (0.00005) [-10.4±0.4%]	0.0101 (0.00004) [-34.9±0.3%]	0.0165 (0.00009) [6.3±0.6%]	0.0368 (0.00008)	0.0263 (0.0001) [-28.6±0.3%]	0.0291 (0.0001) [-21.0±0.4%]	0.0309 (0.0001) [-16.0±0.4%]	0.0284 (0.00007)	0.0244 (0.0001) [-14.1±0.4%]	0.0210 (0.0001) [-26.0±0.4%]	0.0203 (0.00008) [-28.3±0.3%]
ECJRC degradation	Only MultiUse PAs established before 2011*	0.0199 (0.00005)	0.0101 (0.00003) [-49.3±0.2%]	0.0150 (0.00005) [-24.6±0.3%]	0.0179 (0.00008) [-10.1±0.4%]	0.0414 (0.00008)	0.0314 (0.0001) [-24.2±0.3%]	0.0364 (0.0001) [-12.0±0.3%]	0.0375 (0.0001) [-9.4±0.4%]	0.0411 (0.00008)	0.0541 (0.0002) [31.4±0.6%]	0.0329 (0.0001) [-19.8±0.4%]	0.0291 (0.0001) [-29.2±0.3%]
GFW deforestation		0.0400 (0.0001)	0.0498 (0.0003) [24.7±0.8%]	0.0226 (0.0001) [-43.3±0.3%]	0.0286 (0.0001) [-28.5±0.4%]	0.0734 (0.0002)	0.0525 (0.0002) [-28.6±0.3%]	0.0503 (0.0002) [-31.4±0.3%]	0.0497 (0.0002) [-32.3±0.3%]	0.0615 (0.0002)	0.0245 (0.0002) [-60.2±0.3%]	0.0542 (0.0003) [-11.8±0.5%]	0.0339 (0.0002) [-45.0±0.3%]
ECJRC deforestation	PAs established before 2001	0.0127 (0.0001)	0.0151 (0.0002) [19.5±2%]	0.0089 (0.0001) [-29.8±0.9%]	0.0150 (0.0001) [18.8±1.4%]	0.0280 (0.0006)	0.0209 (0.0004) [-25.1±2.1%]	0.0218 (0.0003) [-22.1±2%]	0.0251 (0.0006) [-10.1±2.9%]	0.0185 (0.0002)	0.0149 (0.0001) [-19.4±1.1%]	0.0157 (0.0002) [-14.8±1.3%]	0.0119 (0.0001) [-35.8±0.8%]
ECJRC degradation		0.0210 (0.0005)	0.0195 (0.003) [-7.1±12.8%]	0.0161 (0.0001) [-23.2±2%]	0.0194 (0.00005) [-7.5±2.3%]	0.0407 (0.0006)	0.0369 (0.0001) [-9.5±1.5%]	0.0354 (0.0007) [-13.0±2.2%]	0.0416 (0.001) [2.1±3%]	0.0860 (0.009)	0.07 (0.0006) [-18.6±8.9%]	0.0425 (0.004) [-50.6±7%]	0.0214 (0.0003) [-75.0±2.7%]
GFW deforestation		0.0287 (0.0002)	0.0247 (0.0002) [-13.7±1.1%]	0.0182 (0.0003) [-36.4±1.2%]	0.0255 (0.0004) [-10.9±1.7%]	0.0492 (0.0004)	0.0295 (0.0003) [-40.0±0.9%]	0.0323 (0.0004) [-34.4±1%]	0.0324 (0.0003) [-34.1±0.8%]	0.0337 (0.0004)	0.0149 (0.0002) [-55.9±0.9%]	0.0375 (0.0004) [11.2±1.9%]	0.0171 (0.0002) [-49.3±0.9%]

Supplementary Table 3. Estimated deforestation rates using GFW and ECJRC data from 2010 to 2019 (or 2018 for GFW data) for the country analysis at tropical moist forest extents, standard error in brackets, and percent difference and percent standard error from non-protected area in square brackets.

			GFW deforestation rates				**ECJRC deforestation rates**			
ISO3	**Country**	**N Pixels**	**Non**	**PAs**	**ILs**	**PIAs**	**Non**	**PAs**	**ILs**	**PIAs**
BDI	Burundi	1376	0.051 (0.003)	0.249 (0.037) [389.7±77.2%]	0.033 (0.0005) [-36.0±3.3%]	0.006 (0.0007) [-88.1±1.6%]	0.022 (0.0029)	0.273 (0.038) [1155.7±241.9 %]	0.009 (0.0002) [-57.2±5.8%]	0.003 (0.0004) [-85.2±2.7%]
CAF	Central African Republic	52972	0.011 (0.00008)	0.014 (0.0001) [25.4±1.5%]	0.009 (0.00006) [-14.9±0.9%]	0.008 (0.0001) [-29.2±1.2%]	0.011 (0.00007)	0.014 (0.0001) [33.8±1.5%]	0.011 (0.00008) [4.0±1.1%]	0.007 (0.00007) [-36.7±0.8%]
CMR	Cameroon	125491	0.013 (0.00006)	0.002 (0.00002) [-83.1±0.2%	0.010 (0.00004) [-22.0±0.4%]	0.003 (0.00002) [-80.0±0.2%]	0.006 (0.00003)	0.001 (0.00001) [-80.9±0.2%]	0.004 (0.00003) [-30.5±0.6%]	0.001 (0.00001) [-84.3±0.2%]
COD	Democratic Republic of the Congo	849570	0.024 (0.00005)	0.017 (0.00006) [-28.5±0.3%]	0.016 (0.00003) [-31.6±0.2%]	0.023 (0.00008) [-2.5±0.4%]	0.011 (0.00002)	0.007 (0.00002) [-34.2±0.2%]	0.008 (0.00001) [-28.0±0.2%]	0.013 (0.00005) [24.6±0.5%]
COG	Republic of Congo	134480	0.036 (0.0002)	0.009 (0.0001) [-73.8±0.4%]	0.033 (0.0001) [-9.2±0.6%]	0.023 (0.0003) [-36.5±1%]	0.013 (0.00001)	0.005 (0.00006) [-62.2±0.6%]	0.009 (0.00005) [-26.8±0.8%]	0.004 (0.00007) [-67.2±0.7%]
ETH	Ethiopia	67333	0.041 (0.0002)	0.054 (0.0002) [32.7±1%]	0.034 (0.0002) [-16.1±0.7%]	0.059 (0.0005) [45.9±1.6%]	0.046 (0.0002)	0.068 (0.0005) [46.0±1.3%]	0.041 (0.0001) [-11.9±0.5%]	0.073 (0.0006) [58.1±1.5%]
GAB	Gabon	190579	0.006 (0.00003)	0.007 (0.00009) [16.1±1.6%]	0.008 (0.00003) [32.5±0.8%]	0.003 (0.00003) [-52.3±0.5%]	0.001 (0.000005)	0.001 (0.000008) [2.0±1%]	0.001 (0.000006) [34.2±1%]	0.001 (0.00001) [10.8±1.5%]
KEN	Kenya	31062	0.122 (0.0008)	0.065 (0.001) [-46.7±1.2%]	0.145 (0.001) [19.0±1.2%]	0.127 (0.002) [4.4±1.5%]	0.144 (0.0025)	0.178 (0.0072) [22.9±5.4%]	0.194 (0.0029) [34.5±3%]	0.075 (0.0017) [-48.0±1.5%]

NGA	Nigeria	29393	0.215 (0.001)	0.264 (0.002) [22.9±1%]	0.134 (0.003) [-37.4±1.3%]	0.132 (0.006) [-38.7±2.6%]	0.115 (0.0007)	0.155 (0.0009) [35.5±1.1%]	0.061 (0.0015) [-46.8±1.3%]	0.061 (0.0038) [-46.5±3.4%]
TZA	Tanzania	26468	0.152 (0.0009)	0.083 (0.0006) [-45.4±0.5%]	0.012 (0.001) [-92.4±0.7%]	0.073 (0.01) [-52.1±6.3%]	0.004 (0.0002)	0.002 (0.0001) [-42.6±4.8%]	0.001 (0.00005) [-74.9±.9%]	0.007 (0.0010) [68.6±25.6%]
UGA	Uganda	4459	0.170 (0.003)	0.111 (0.004) [-34.9±2.5%]	0.102 (0.006) [-40.0±3.5%]	0.062 (0.004) [-63.8±2.4%]	0.054 (0.0009)	0.029 (0.0011) [-45.8±2.2%]	0.055 (0.0015) [3.2±3.2%]	0.027 (0.0010) [-49.2±2%]
BLZ	Belize	10258	0.208 (0.003)	0.123 (0.004) [-40.9±2.3%]	0.350 (0.003) [68.6±2.6%]	0.125 (0.003) [-39.8±1.8%]	0.139 (0.0011)	0.096 (0.0018) [-30.5±1.4%]	0.096 (0.0012) [-30.9±1%]	0.063 (0.0025) [-54.2±1.8%]
BOL	Bolivia	433265	0.068 (0.0002)	0.046 (0.0001) [-32.9±0.3%]	0.038 (0.00008) [-44.9±0.2%]	0.035 (0.0001) [-48.2±0.2%]	0.038 (0.0001)	0.043 (0.0002) [13.1±0.5%]	0.081 (0.0002) [113.7±0.9%]	0.046 (0.0002) [20.2±0.7%]
BRA	Brazil	1049326	0.043 (0.00005)	0.024 (0.00004) [-44.2±0.1%]	0.019 (0.0001) [-52.3±0.3%]	0.023 (0.00004) [-45.6±0.1%]	0.018 (0.00002)	0.011 (0.00002) [-38.1±0.1%]	0.010 (0.00005) [-45.6±0.3%]	0.016 (0.00003) [-15.8±0.2%]
COL	Colombia	402454	0.058 (0.0001)	0.028 (0.00009) [-52.0±0.2%]	0.035 (0.0001) [-39.8±0.2%]	0.041 (0.0002) [-29.1±0.4%]	0.026 (0.00006)	0.016 (0.0005) [-39.2±0.2%]	0.013 (0.00005) [-49.6±0.2%]	0.017 (0.0001) [-37.0±0.4%]
CRI	Costa Rica	13205	0.037 (0.0003)	0.023 (0.0004) [-37.9±1.1%]	0.054 (0.0004) [46.3±1.7%]	0.019 (0.0007) [-47.6±1.9%]	0.034 (0.0003)	0.034 (0.0005) [-2.3±1.6%]	0.062 (0.0009) [78.7±3%]	0.050 (0.0024) [46.4±7%]
ECU	Ecuador	63810	0.039 (0.0002)	0.019 (0.0001) [-52.0±0.4%]	0.032 (0.0001) [-19.4±0.4%]	0.008 (0.00007) [-82.8±0.2%]	0.054 (0.0003)	0.039 (0.0004) [-28.5±0.7%]	0.029 (0.0001) [-46.6±0.4%]	0.019 (0.0003) [-65.5±0.6%]
GTM	Guatemala	108132	0.130 (0.0008)	0.216 (0.001) [65.6±1.5%]	0.190 (0.0005) [45.6±1%]	0.230 (0.002) [76.6±1.6%]	0.034 (0.0002)	0.077 (0.0004) [125.4±1.8%]	0.045 (0.0001) [31.5±0.9%]	0.079 (0.0005) [133.8±2.1%]
GUF	French Guiana	46348	0.004 (0.00008)	0.011 (0.0003) [168.7±8.4%]	0.0003 (0.0002) [-93.1±4.9%]	0.017 (0.0004) [294.9±12.6%]	0.002 (0.00002)	0.003 (0.00008) [63.8±4.3%]	0.0002 (0.00004) [-90.1±1.9%]	0.004 (0.00007) [120.2±4%]
GUY	Guyana	58900	0.006	0.001 (0.00001)	0.024 (0.0003)	0.007 (0.00009)	0.008	0.0006	0.017 (0.0002)	0.008 (0.0001)

			(0.00008)	[-91.3±0.2%]	[285.9±7.3%]	[17.7±2.2%]	(0.00008)	(0.000009) [-92.9±0.1%]	[109.6±2.7%]	[-1.9±1.5%]
HND	Honduras	53855	0.108 (0.0003)	0.105 (0.0006) [-2.7±0.6%]	0.060 (0.0002) [-44.7±0.3%]	0.080 (0.0009) [-25.7±0.8%]	0.033 (0.0001)	0.038 (0.0002) [17.1±0.9%]	0.0178 (0.00009) [-45.4±0.4%]	0.028 (0.00003) [-14.2±0.9%]
MEX	Mexico	636165	0.047 (0.0001)	0.025 (0.0001) [-47.9±0.3%]	0.079 (0.0002) [66.5±0.5%]	0.049 (0.0004) [3.8±0.9%]	0.010 (0.00003)	0.010 (0.00005) [8.2±0.6%]	0.008 (0.00002) [-18.1±0.3%]	0.007 (0.00006) [-28.5±0.7%]
NIC	Nicaragua	58792	0.111 (0.0007)	0.133 (0.0005) [20.0±0.9%]	0.186 (0.0016) [68.1±1.8%]	0.241 (0.0017) [117.4±2.1%]	0.054 (0.0004)	0.060 (0.0003) [10.9±1%]	0.089 (0.0009) [63.2±2.2%]	0.128 (0.0014) [135.5±3.2%]
PAN	Panama	34046	0.119 (0.0007)	0.047 (0.0007) [-60.5±0.6%]	0.101 (0.0005) [-14.9±0.7%]	0.084 (0.0012) [-29.8±1.1%]	0.064 (0.0005)	0.033 (0.0005) [-47.9±0.9%]	0.055 (0.0004) [-14.7±0.9%]	0.042 (0.0005) [-34.6±0.9%]
PER	Peru	658079	0.038 (0.0001)	0.010 (0.00003) [-74.2±0.1%]	0.029 (0.00009) [-24.1±0.3%]	0.005 (0.00007) [-86.7±0.2%]	0.020 (0.00005)	0.009 (0.00002) [-55.5±0.2%]	0.013 (0.00004) [-33.7±0.3%]	0.002 (0.00004) [-90.9±0.2%]
PRY	Paraguay	11753	0.219 (0.0023)	0.084 (0.0038) [-61.5±1.8%]	0.127 (0.0019) [-42.0±1.1%]	0.085 (0.0012) [-61.1±0.7%]	0.060 (0.0004)	0.023 (0.0005) [-60.9±0.8%]	0.071 (0.0004) [18.9±1.1%]	0.036 (0.0002) [-39.9±0.5%]
SUR	Suriname	129099	0.008 (0.0003)	0.0005 (0.00001) [-93.7±0.3%]	0.037 (0.0007) [346.9±19.4%]	0.0007 (0.00002) [-91.0±0.5%]	0.008 (0.00005)	0.002 (0.0004) [-72.7±0.5%]	0.010 (0.00006) [17.4±1%]	0.003 (0.00006) [-63.6±0.7%]
VEN	Venezuela	479285	0.012 (0.00003)	0.013 (0.00004) [5.7±0.4%]	0.018 (0.00007) [40.7±0.7%]	0.020 (0.00004) [60.7±0.5%]	0.021 (0.00008)	0.016 (0.00003) [-23.2±0.3%]	0.043 (0.0003) [103.9±1.6%]	0.028 (0.00007) [31.9±0.6%]
AUS	Australia	9890	0.017 (0.0004)	0.041 (0.0021) [135.0±13.1%]	0.012 (0.0003) [-28.2±2.2%]	0.009 (0.0004) [-48.9%]	0.011 (0.0003)	0.014 (0.0008) [28.0±7.8%]	0.005 (0.00007) [-58.8±1.2%]	0.003 (0.00009) [-71.3±1.1%]
BGD	Bangladesh	6603	0.106 (0.0024)	0.123 (0.0039) [16.1±5.3%]	0.117 (0.0015) [10.2±2.9%]	0.120 (0.0022) [13.3±3.3%]	0.017 (0.0005)	0.020 (0.0009) [18.7±6.4%]	0.009 (0.0002) [-48.2±1.9%]	0.012 (0.0007) [-26.2±4.6%]

IDN	Indonesia	645788	0.053 (0.00009)	0.036 (0.0002) [-33.3±0.3%]	0.045 (0.00006) [-16.4±0.2%]	0.027 (0.00007) [-48.60.2%]	0.023 (0.00004)	0.021 (0.00006) [-11.4±0.3%]	0.019 (0.00003) [-18.9±0.2%]	0.012 (0.00004) [-47.6±0.2%]
IND	India	536774	0.015 (0.00006)	0.007 (0.0001) [-50.9±0.8%]	0.047 (0.0001) [211.3±1.5%]	0.017 (0.0003) [13.8±1.7%]	0.005 (0.00002)	0.004 (0.00003) [-29.5±0.6%]	0.002 (0.000006) [-56.6±0.2%]	0.001 (0.00001) [-83.4±0.3%]
KHM	Cambodia	118054	0.329 (0.0010)	0.225 (0.0011) [-31.7±0.4%]	0.287 (0.0009) [-13.0±0.4%]	0.230 (0.0018) [-30.1±0.6%]	0.066 (0.0003)	0.054 (0.0004) [-18.8±0.7%]	0.058 (0.0002) [-11.7±0.6%]	0.045 (0.0005) [-32.4±0.8%]
LAO	Laos	231302	0.150 (0.0006)	0.028 (0.0004) [-81.2±0.3%]	0.225 (0.0003) [50.3±0.6%]	0.108 (0.0005) [-27.9±0.4%]	0.064 (0.0002)	0.047 (0.0003) [-28.1±0.5%]	0.070 (0.00007) [8.1±0.3%]	0.058 (0.0002) [-9.4±0.4%]
LKA	Sri Lanka	13823	0.063 (0.0008)	0.024 (0.0003) [-61.2±0.8%]	0.031 (0.0025) [-50.7±4%]	0.011 (0.0012) [-82.4±2%]	0.019 (0.0002)	0.018 (0.0002) [-4.1±1.3%]	0.005 (0.0002) [-75.9±1.3%]	0.029 (0.0004) [54.9±2.6%]
MMR	Myanmar	422123	0.062 (0.0002)	0.059 (0.0015) [-5.9±2.4%]	0.096 (0.0001) [53.8±0.5%]	0.029 (0.0001) [-54.1±0.2%]	0.022 (0.00006)	0.022 (0.0003) [0.0±1.3%]	0.024 (0.00003) [9.5±0.3%]	0.033 (0.00007) [51.6±0.5%]
MYS	Malaysia	217682	0.106 (0.0002)	0.019 (0.0002) [-82.1±0.2%]	0.140 (0.0002) [32.3±0.3%]	0.018 (0.00008) [-82.7±0.1%]	0.024 (0.00007)	0.005 (0.00006) [-81.7±0.2%]	0.030 (0.00005) [23.6±0.4%]	0.008 (0.00003) [-69.2±0.2%]
NCL	New Caledonia	9071	0.013 (0.0002)	0.011 (0.0005) [-15.0±4.2%]	0.011 (0.0002) [-12.0±1.7%]	0.003 (0.00009) [-74.0±0.8%]	0.030 (0.0004)	0.031 (0.0004) [2.2±1.8%]	0.039 (0.0009) [28.0±3.5%]	0.030 (0.0009) [0.1±3.2%]
NPL	Nepal	25846	0.009 (0.0001)	0.024 (0.0009) [174.4±10.5%]	0.006 (0.0001) [-27.8±1.7%]	0.008 (0.0005) [-12.3±5.9%]	0.002 (0.00006)	0.005 (0.0006) [110.2±5.9%]	0.002 (0.00002) [-37.5±1.9%]	0.004 (0.0005) [59.6±19.7%]
PHL	Philippines	109286	0.056 (0.0002)	0.056 (0.0002) [-0.5±0.5%]	0.063 (0.0002) [13.5±0.6%]	0.055 (0.0005) [-0.6±1%]	0.035 (0.0001)	0.026 (0.00009) [-25.1±0.4%]	0.047 (0.0002) [35.3±0.7%]	0.039 (0.0003) [10.0±1%]
THA	Thailand	379883	0.101	0.048 (0.0005)	0.089 (0.0002)	0.038 (0.0004)	0.004	0.004 (0.00002)	0.004	0.005 (0.00003)

			(0.0004)	[-53.0±0.6%]	[-12.7±0.4%]	[-62.2±0.4%]	(0.00001)	[5.8±0.6%]	(0.000006) [-2.0±0.3%]	[22.4±0.7%]
VNM	Vietnam	213560	0.131 (0.0004)	0.039 (0.0004) [-70.6±0.3%]	0.136 (0.0002) [3.3±0.4%]	0.047 (0.0005) [-64.3±0.4%]	0.044 (0.0001)	0.029 (0.0003) [-33.9±0.6%]	0.043 (0.00006) [-1.1±0.3%]	0.030 (0.0002) [-31.6±0.5%]

Supplementary Table 4. Data used, description, source and rationale. Name of variable as used in the analysis is showed in fixed width font.

Variable	**Description**	**Source**	**Rationale**
Deforested, degraded, and TMF forest pixels (`Deforested2018, Degraded2018, DeforestedDegraded2018, TMF2018`)	No. of 30 m pixels listed as deforested, degraded between 2010 and 2019, and all tropical moist forest pixels	European Commission Joint Research Centre Long-term monitoring of tropical moist forest extents Subtypes data. Vancutsem et al., 2021 https://forobs.jrc.ec.europa.eu/TMF/download/	Outcome
Forested Pixels (`ForestedPixels2010, ForestedPixels2018`)	No. of 30 m pixels with ≥25% tree canopy cover (min 0, max 1024) in 2010 & 2018 within ~1 km pixel	GLAD, University of Maryland, 2013; Global Forest Watch, 2018 Tree cover 2010: https://glad.umd.edu/Potapov/TCC_2010 Datamask and loss year (v1.6): https://earthenginepartners.appspot.com/science-2013-global-forest/download_v1.6.html	Outcome
Protection Type (`Type`)	Type of forest protection: None, PAs, ILs, or PIAs	UNEP-WCMC and IUCN, 2020; Garnett et al., 2018	Treatment
Slope (`slope`)	Mean slope (degrees) in ~1 km pixel	Amatulli et al., 2018 http://www.earthenv.org/topography	Affects likelihood of deforestation & PA location
Elevation (`elevation`)	Mean elevation (metres) in ~1 km pixel	Amatulli et al., 2018 http://www.earthenv.org/topography	Affects likelihood of deforestation & PA location
Population Density (`ppp`)	Mean number of people in a ~1 km pixel in 2010	Lloyd et al., 2017 https://www.worldpop.org/geodata/listing?id=64	Affects likelihood of deforestation
Travel Time (`travelTimeto5kcity`)	Travel time to nearest urban area with at least 5000 inhabitants in 2015 (mins)	Nelson et al., 2019 https://figshare.com/articles/Travel_time_to_cities_and_ports_in_the_year_2015/7638134/3	Affects likelihood of deforestation

Distance To Roads (`DisttoRoads`)	Euclidean distance to nearest road (metres)	CIESIN, Columbia University, 2013; Euclidean distance calculated by authors using ArcGIS https://sedac.ciesin.columbia.edu/data/set/groads-global-roads-open-access-v1/data-download	Affects likelihood of deforestation
Forest Area (`Forest2010Area, Forest2000Area`)	Area in ~1 km pixel that was forested in 2010 or 2000 at 25% tree canopy threshold (m^2)	Global Forest Watch, 2018; We multiplied the number of 30 m pixels in each 1 km cell that were forested in 2000 or 2010 by the mean 30 m-pixel area using the area function in the raster package (Hijmans, 2020) to obtain baseline forest area. Tree cover 2010: https://glad.umd.edu/dataset/global-2010-tree-cover-30-m Tree cover 2000: https://earthenginepartners.appspot.com/science-2013-global-forest/download_v1.6.html	Affects likelihood of deforestation
Country (`gadm36`)	Country boundaries	Global Administrative Areas, 2018 (v3.6) https://gadm.org	Control for country-level political and economic effects
Spatial Database of Planted Trees	Vector data of planted trees (including forest plantations of native or introduced species and agricultural tree crops) for 82 countries	Global Forest Watch, 2019 https://data.globalforestwatch.org/documents/gfw::planted-forests/about	Used to mask out known tree plantations from study area
Tree Plantations	Vector data of tree plantations for Brazil, Cambodia, Indonesia, Liberia, Malaysia and Peru	Transparent World and Global Forest Watch, 2019 https://data.globalforestwatch.org/datasets/gfw::tree-plantations/explore	Used to mask out known tree plantations from study area

Supplementary Table 5. Number of pixels (including control and treatment) and mean value of standardised difference in means (SMD) across the six covariates before and after matching with propensity score at the regional level for PAs established before 2011 at tropical moist forest extents for the five samples.

		Africa			Americas			Asia-Pacific		
		PA	**IL**	**PIA**	**PA**	**IL**	**PIA**	**PA**	**IL**	**PIA**
Complete Dataset	N pixels	2920015	3121726	2688449	9394688	8802762	9162886	3236908	5302259	3386464
	mean SMD	0.3905	0.2778	0.4483	0.4439	0.3559	1.2761	0.4795	0.5893	1.934
Sample 1	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.3883	0.2786	0.4478	0.4383	0.3594	1.2528	0.468	0.6601	2.0261
After matching	N pixels	95096	91402	93678	105996	86336	73942	95184	56382	71238
	mean SMD	0.0651	0.0768	0.1106	0.0764	0.2907	0.1612	0.1468	0.2443	0.2732
Sample 2	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.4013	0.2776	0.4445	0.4389	0.3535	1.4228	0.4646	0.5656	1.885
After matching	N pixels	95972	90726	94468	105894	85678	73158	95232	56304	70552
	mean SMD	0.0659	0.0775	0.1112	0.0758	0.2884	0.1601	0.1463	0.2357	0.2752
Sample 3	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.3994	0.2814	0.4471	0.4385	0.3618	1.324	0.5009	0.6151	1.8664
After matching	N pixels	95926	90994	94432	105844	85414	73416	95696	57504	71206
	mean SMD	0.071	0.0767	0.1108	0.0728	0.296	0.1622	0.1489	0.2409	0.2708
Sample 4	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104

	mean SMD	0.4246	0.2768	0.4413	0.4451	0.35	1.2751	0.5361	0.613	2.048
After matching	N pixels	95356	91330	94468	105722	85732	73534	94864	56500	71448
	mean SMD	0.0709	0.0731	0.1104	0.0751	0.2934	0.169	0.1516	0.2423	0.2818
Sample 5	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.416	0.2751	0.4582	0.4515	0.3531	1.3648	0.476	0.5875	1.8777
After matching	N pixels	95380	90984	93738	105756	85318	73152	95388	57084	71612
	mean SMD	0.0705	0.0766	0.1114	0.0776	0.2946	0.156	0.1456	0.2376	0.259

Supplementary Table 6. Number of pixels (including control and treatment) and mean value of standardised difference in means (SMD) across the six covariates before and after matching with propensity score for the 42 countries. Red font refers to cases where matching with replacement was done. Please refer to Supplementary Table 3 for country names from ISO3 code.

	PA		IL		PIA	
Countries	**N pixels**	**mean SMD**	**N pixels**	**mean SMD**	**N pixels**	**mean SMD**
BDI	10979	1.1247	11501	1.1124	11169	6.4028
After matching	176	0.1282	1206	0.1224	30	0.8781
CAF	47405	0.4039	52038	0.397	45757	0.6224
After matching	18796	0.0428	29568	0.0311	12940	0.071
CMR	202539	0.6898	245942	0.2226	192940	2.6319
After matching	25368	0.0931	97234	0.0503	10082	0.0907
COD	929404	0.3093	1097108	0.1454	883748	0.1767
After matching	230222	0.0242	561266	0.0235	138910	0.0259
COG	185212	0.5141	212650	0.3222	167534	0.3893
After matching	39530	0.0779	95324	0.0458	4930	0.0569
ETH	13194	0.4178	62395	0.2285	14768	0.3627
After matching	4718	0.0789	61085	0.0771	6676	0.0531
GAB	146123	0.2901	180352	0.1929	128505	0.3539
After matching	57204	0.0633	130032	0.0292	36240	0.0552
KEN	40205	1.1628	53079	0.5743	43497	0.9041

After matching	3984	0.1021	22770	0.0811	9494	0.086
NGA	124098	1.0916	115662	1.3526	109441	2.5265
After matching	27378	0.0875	2020	0.1703	290	0.527
TZA	83086	0.7079	65084	1.738	65216	2.0053
After matching	25690	0.0731	616	0.103	228	0.3085
UGA	26132	1.4833	23145	0.7485	24118	1.2103
After matching	2204	0.1838	1726	0.0966	800	0.1457
BLZ	10157	1.3221	9905	0.2329	6811	1.3333
After matching	2670	0.0834	7372	0.0426	1540	0.1626
BOL	473519	0.2902	458092	0.7632	410269	1.7419
After matching	214340	0.0532	193302	0.0337	95812	0.0567
BRA	1484215	0.6013	4402983	1.8028	1477989	1.6408
After matching	579744	0.0794	51966	0.109	471116	0.0923
COL	683236	0.512	858931	3.8467	625870	4.2561
After matching	182126	0.0255	235330	0.2133	57068	0.4186
CRI	43399	1.6389	39482	1.789	36866	4.7932
After matching	8190	0.134	5932	0.2304	642	0.7411
ECU	152168	1.6618	189635	1.6892	144602	2.8258
After matching	20076	0.1294	44752	0.2207	5500	0.4901
GTM	44332	2.0149	94042	0.4564	37244	0.3489
After matching	23600	0.1443	83879	0.0651	13708	0.0517

GUF	82888	0.5361	43380	1.3306	50413	0.8713
After matching	41084	0.1184	668	0.0974	7456	0.0746
GUY	183101	3.8689	197687	0.6011	185798	10.3796
After matching	8890	0.3119	37826	0.0236	13654	1.0065
HND	80363	0.9176	85181	0.2294	75919	1.2225
After matching	16376	0.077	36220	0.0503	8328	0.1441
MEX	706499	0.2508	866945	0.2325	652050	0.5565
After matching	155930	0.0573	481704	0.0368	53282	0.167
NIC	84548	0.7196	69555	0.4825	72628	1.9185
After matching	42244	0.0377	18180	0.0654	6034	0.309
PAN	42879	2.0197	60376	1.2348	48602	5.7735
After matching	6070	0.1707	27106	0.1069	6330	0.7308
PER	662861	0.6576	714747	0.2605	533436	1.2916
After matching	300328	0.0534	405090	0.0304	42468	0.0509
PRY	97511	1.1524	100709	0.7977	98522	2.102
After matching	1382	0.1697	7862	0.0681	2826	0.1442
SUR	92838	0.7926	133685	0.1307	88403	0.4894
After matching	22752	0.1961	105030	0.0309	14466	0.1645
VEN	228108	0.4439	266717	0.7223	415776	2.665
After matching	146920	0.091	92160	0.0893	309240	0.2599

AUS	14542	2.5269	17380	0.504	14350	3.7927
After matching	2122	0.3458	7646	0.0258	1028	0.5919
BGD	111039	0.614	125162	2.0606	111301	2.2974
After matching	1006	0.1149	5438	0.2507	552	0.2291
IDN	761114	1.0132	597087	1.0441	875068	2.5413
After matching	81340	0.0704	421715	0.0599	196584	0.154
IND	679762	0.4317	954236	0.9942	669113	1.6759
After matching	54236	0.0375	485058	0.1152	32938	0.1159
KHM	100529	1.2065	119012	0.4091	97171	2.0849
After matching	38922	0.0743	82386	0.0579	24332	0.175
LAO	51320	1.5417	220977	0.7795	84208	1.8312
After matching	8610	0.0589	206453	0.0912	38020	0.196
LKA	54399	1.0887	43787	1.0103	43928	3.4129
After matching	13326	0.0976	268	0.1094	460	0.2551
MMR	238399	1.0825	468922	0.6592	262377	1.8945
After matching	11050	0.0658	383221	0.1131	45980	0.0843
MYS	60598	2.9546	217299	0.6982	70868	2.2527
After matching	7832	0.1696	201440	0.073	22352	0.1129
NCL	10092	0.3105	12736	0.8067	8605	1.0563
After matching	3736	0.0309	5518	0.0761	762	0.1179

NPL	10156	1.0149	27510	0.5179	8897	0.9606
After matching	2246	0.0926	23772	0.0639	174	0.0937
PHL	208409	0.7228	199350	1.0371	183432	1.2307
After matching	63558	0.0428	45328	0.0497	13600	0.0503
THA	223873	2.6304	413734	0.2354	251935	2.9363
After matching	29580	0.218	340644	0.0303	38708	0.3096
VNM	143486	0.9394	295319	1.2779	148846	3.0668
After matching	10358	0.0575	197860	0.1356	16178	0.1238

Supplementary Methods

Spatial data cleaning and manipulation were done in R 3.6.2, QGIS 3.4 and ArcMap 10.7.1. The following R packages were used: raster (v3.0.12; Hijmans, 2020), rgdal (v1.4.8; Bivand et al., 2020), rgeos (v0.5.2; Bivand & Rundel, 2020), sf (v0.9.6; Pebesma, 2018), fasterize (v1.0.0; Ross, 2020), and wdpar (v1.0.4; Hanson, 2020).

Data

Protected Areas

We downloaded the January 2020 version of World Database of Protected Areas using the `wdpar` package's `wdpa_fetch('global')` function on 16 January 2020. Following the recommended protocol for cleaning data, we removed non-established sites, UNESCO man and biosphere sites, and point sites, using the source code of `wdpa_clean` function from `wdpar` package. Reprojected the files to ESRI 54009, and intersected WDPA with Tropical Forest extent to retain only PAs that fall within tropical forests. We filtered out marine PAs and edited a few features (Supplementary Table 8) to obtain a 'flat' layer of WDPA which has no overlapping polygons in R. We subset PAs that were established before the year 2011 and before the year 2001 for the precautionary analysis; PAs that had no establishment year (value of 0 in the STATUS_YEAR column) were assumed to have been established before the study period and were included in both analyses.

Supplementary Table 7. WDPA features edited to obtain a 'flat' layer

WDPAID	Error	Action
555624205	Self-intersection at -6026876, 2010412.	Deleted the part of feature that crossed another
957	Too few points at 2971940, -68489	Deleted vertices of the feature that overlapped another
145541	Self-intersection at -9120493, 1907430	Deleted this feature which was identical to WDPAID 478291
478291	Self-intersection at 2969095, -88786	Deleted this feature which was identical to WDPAID 2017
60, 12203, 15	Too few points at -5106215, -3120100	Moved vertices of these features so they don't cross

To obtain PA-only areas, we filtered out PAs that were governed by Indigenous people as listed in the attribute table, GOV_Type column, and used the `erase` tool in ArcMap to remove areas of the PIA spatial intersection.

Our spatial intersect of Protected-Indigenous Areas reveal that most PIAs do not fall within multi-use PAs (Supplementary Table 8), though the proportion of multi-use PAs has increased over time.

IUCN management categories might have implications for deforestation, we considered the case that multi-use PAs (categories V and VI) might differ from strict PAs (categories I-IV). We filtered for multi-use PAs and conducted a separate analysis for those.

To ensure we do not include areas that were subsequently designated as PAs, we identified PAs that were established from 2011 or from 2001 onwards (for the respective analysis) and rasterised them in R to mask these areas out from further analysis.

Supplementary Table 8. IUCN management categories within our datasets

PA management category	**Protected Areas (before 2011)**	**Protected-Indigenous Areas (before 2011)**	**Protected Areas (before 2001)**	**Protected-Indigenous Areas (before 2001)**
Ia – Strict Nature Reserve	270	163	223	156
Ib – Wilderness Area	93	2	72	2
II – National Park	745	326	518	250
III – Natural Monument or Feature	87	26	44	23
IV – Habitat/species Management Area	545	205	422	169
Strict PAs (% of all PAs)	**44.0**	**41.0**	**44.7**	**49.7**
V – Protected landscape/seascape	386	103	268	64
VI – Protected Area with Sustainable Use of Natural Resources	553	224	319	113
Multi-Use PAs (% of all PAs)	**23.7**	**18.6**	**20.5**	**14.7**
Not applicable	29	13	27	11

Not assigned	4	3	2	1
Not reported	1243	697	966	419

Indigenous Lands

We obtained the geospatial layer for Indigenous Peoples Land (IPL) from Garnett et al 2018. We selected IPLs that intersected tropical forest biome using ArcMap `Select Layer by Location` tool, `intersect` function.

To obtain IL-only areas, we used the `erase` tool in ArcMap to remove areas of PIA.

Protected and Indigenous Areas (PIA)

We used the `intersect` tool in QGIS to intersect layer of WDPA that is not governed by Indigenous People with layer of IPL, and used Dissolve tool to dissolve the subsequent layer by WDPAID such that PAs which were originally one still remain as one feature. We joined this layer with the layer of WDPA that is governed by Indigenous People to obtain a 'PIA' layer.

ECJRC Tropical Moist Forest deforestation and degradation

We used the Subtypes dataset. We defined deforested pixels as those where deforestation started in 2010-2016 and included recent degradation/deforestation from 2017-2019 (classes 42, 51, 52, 53 and 54), and degraded pixels as those where degradation started in 2010-2016 and included recent degradation/deforestation from 2017-2019 (classes 22, 24, 26, 51, 52, 53 and 54). We defined tropical moist forest pixels as those not classed as permanent water (class 71) at the start of the monitoring period. We additionally considered deforested and degraded pixels together (classes 22, 24, 26, 42, 51, 52, 53 and 54). We aggregated the 1 arc-second pixels by 31 to obtain the number of deforested/degraded/TMF 30 m pixels within ~1 km cells (max 961). We used a different aggregation factor to the GFW dataset due to minor discrepancies in the native resolution (0.0002694946, compared to 0.00025 for the GFW data)

Global Forest Watch forest loss

We applied 25% tree cover threshold to define forested pixels, for tree cover 2000 and tree cover 2010. We aggregated 30 m or 1 arc-second pixels by 32 (which approximates to 1 km or 30 arc-seconds) in Python to obtain ~1 km resolution cells of number of forested 30 m pixels in 2000, 2010 and 2018 (max 1024). We calculated mean area of the 30m-pixels in each 32-factor aggregated cell in R using the `area` function of `raster` package, since the datasets are in EPSG

4326 and not equal-area projection. We then multiplied the area by number of forested pixels in 2000 or 2010 to obtain forested area in 2000 or 2010.

Distance to roads

We clipped the road geospatial layer to tropical forest extent, re-projected to EPRI 54009, and rasterised it. Used the `distance` tool, `Euclidean distance planar` in ArcMap to calculate distance from each raster cell to the nearest road raster cell, and re-projected back to EPSG 4326.

Harmonising layers for analysis

All raster layers (forested pixels 2010, forested pixels 2018, degraded pixels in 2018, deforested pixels in 2018, TMF pixels in 2018, protection type, slope, elevation, population density, travel time to nearest urban area, distance to roads, forest area in 2000, forest area in 2010 and countries), were clipped and resampled to tropical forest extent, with the post-2000 or post-2010 PA mask layer applied for the relevant analysis. We additionally masked out known tree plantations. Raster layers were then converted to comma separated value files for use in matching and regression.

Analysis

To conduct the matching, files were split by protection types (non-protected and PA-only, non-protected and IL-only, non-protected and PIA) and IPBES regions. We followed Brooks et al. (2016) convention of sorting overseas territories according to geography rather than governing territory (Supplementary Table 1). For each protection type and IPBES region dataset (e.g. PA and non-protected area pixels within the Americas, or IL and non-protected area pixels within Africa; total of nine datasets), we took five separate samples and ran the matching algorithm for each sample using the MatchIt package (v3.0.2; Ho *et al*. 2018) in R. We included all numeric covariates and specified exact matching for country.

```
myMatch <-
matchit(Type~slope+elevation+ppp+travelTimeto5kcity+DisttoRoads+Forest2010Area,
data=datSample, method='nearest', exact='gadm36', caliper=0.25)
```

The matched data were then merged for each region (e.g. matched non-protected area and PA pixels, matched non-protected area and IL pixels, and matched non-protected area and PIA pixels for Africa) to run regional regression models, with any duplicate non-protected area pixels removed. This was repeated for the five samples. We used the mgcv package (v1.8.31; Wood 2019) to run the generalised additive mixed models. We fitted separate models for ECJRC deforestation and degradation, GFW deforestation, ECJRC deforestation only, and ECJRC degradation only.

```
# formula for ECJRC deforestation and degradation GAMM

gamform_ECJRCdefdeg <- cbind(DeforestedDegraded2018, TMF2018) ~ Type + s(x,y, k=100) + s(slope, bs='cr') + s(elevation, bs='cr') + s(ppp, bs='cr') + s(travelTimeto5kcity, bs='cr') + s(DisttoRoads, bs='cr') + s(Forest2010Area, bs='cr') + s(gadm36, Forest2010Area, bs='re') + s(gadm36, bs='re') + s(Type, gadm36, bs='re')

# formula for GFW deforestation GAMM

gamform_GFWdef <- cbind(ForestedPixels2010-ForestedPixels2018, ForestedPixels2018) ~ Type + s(x,y, k=100) + s(slope, bs='cr') + s(elevation, bs='cr') + s(ppp, bs='cr') + s(travelTimeto5kcity, bs='cr') + s(DisttoRoads, bs='cr') + s(Forest2010Area, bs='cr') + s(gadm36, Forest2010Area, bs='re') + s(gadm36, bs='re') + s(Type, gadm36, bs='re')

# formula for ECJRC deforestation only GAMM

gamform_ECJRCdef <- cbind(Deforested2018, TMF2018) ~ Type + s(x,y, k=100) + s(slope, bs='cr') + s(elevation, bs='cr') + s(ppp, bs='cr') + s(travelTimeto5kcity, bs='cr') + s(DisttoRoads, bs='cr') + s(Forest2010Area, bs='cr') + s(gadm36, Forest2010Area, bs='re') + s(gadm36, bs='re') + s(Type, gadm36, bs='re')

# formula for ECJRC forest degradation GAMM

gamform_ECJRCdeg <- cbind(Degraded2018, TMF2018) ~ Type + s(x,y, k=100) + s(slope, bs='cr') + s(elevation, bs='cr') + s(ppp, bs='cr') + s(travelTimeto5kcity, bs='cr') + s(DisttoRoads, bs='cr') + s(Forest2010Area, bs='cr') + s(gadm36, Forest2010Area, bs='re') + s(gadm36, bs='re') + s(Type, gadm36, bs='re')

# fitting the GAMM model (in this example, for GFW deforestation data)

gam_GFWdef <- bam(gamform_GFWdef, data=df, family=binomial(link='logit'), discrete=TRUE)
```

We then used the models to estimate deforestation (or degradation) rates, using the mean values for numeric covariates, and the same values of x and y coordinates, countries and protection type. The mean value across the five samples was then calculated and presented in the results.

```
# estimate deforestation rates (in this example, for GFW deforestation rates)

# create new X values

newX_GFWdef <- df %>%
```

```
    select(slope, elevation, ppp, travelTimeto5kcity, DisttoRoads,
Forest2010Area) %>%
    summarise_all(mean, na.rm=TRUE) %>%
    data.frame(Type=gam_GFWdef$model$Type,
               gadm36=gam_GFWdef$model$gadm36,
               x=gam_GFWdef$model$x,
               y=gam_GFWdef$model$y)

# predict new Y values
newY_GFWdef <- predict(gam_GFWdef, newdata=newX_GFWdef, type='link',
se.fit=TRUE)
newY_GFWdef <- as.data.frame(newY_GFWdef) %>%
    mutate(fitProb = boot::inv.logit(fit),
           lwr = fit - se.fit,
           upr = fit + se.fit,
           lwrProb = boot::inv.logit(lwr),
           uprProb = boot::inv.logit(upr))

# estimated deforestation rates
newType_GFWdef <- data.frame(newX_GFWdef, newY_GFWdef$fitProb,
newY_GFWdef$lwrProb, newY_GFWdef$uprProb) %>%
    dplyr::rename(fitProb = newY_GFWdef.fitProb,
                  lwr = newY_GFWdef.lwrProb,
                  upr = newY_GFWdef.uprProb)

# calculate mean estimated deforestation rates for each region and protection
type for each sample
postGAMDR_GFWdef <- newType_GFWdef %>%
    group_by(Type) %>%
    dplyr::summarise(mean = mean(fitProb),
                     n = n(),
                     sd = sd(fitProb),
                     se = sd(fitProb)/sqrt(n()),
                     lwr = mean(lwr),
                     upr = mean(upr)) %>%
    mutate(Process = "PostGAMMDR",
           Dataset = "GFW-DefRate")

# summarised values across the five samples
```

```
dat2 <- dat %>%
    mutate(Type = as.factor(Type),
           SampleNumber = as.factor(SampleNumber),
           Region = as.factor(Region),
           Process = as.factor(Process),
           Dataset = as.factor(Dataset)) %>%
    filter(Process == 'PostGAMMDR') %>%
    group_by(Dataset, Type, Region) %>%
    summarise(y = mean(mean),
              y_sd = sd(mean),
              y_se = sd(mean)/sqrt(n()),
              n = n())
```

For the country-level analysis, the data were split by protection types and countries (for the 42 countries which had PA, IL, and PIA pixels), and matching was done for each protection type and country dataset (i.e. 126 datasets, for example, of non-protected area pixels and PA pixels in Colombia). Samples were only taken for large datasets with more than 300,000 treated pixels (i.e. PAs in Brazil, ILs in Indonesia, ILs in Myanmar, and PIAs in Brazil), sampling down to about 300,000 treated pixels within the dataset. Matching was done with replacement if the number of treated pixels (PA, IL, or PIA pixels) exceeded the number of control pixels (non-protected area pixels), and without replacement otherwise.

```
# matching with replacement

myMatch <-
matchit(Type~slope+elevation+ppp+travelTimeto5kcity+DisttoRoads+Forest2010Area,d
ata=dat, method='nearest', replace=TRUE, caliper=0.25)

# matching without replacement

myMatch <-
matchit(Type~slope+elevation+ppp+travelTimeto5kcity+DisttoRoads+Forest2010Area,
data=dat, method='nearest', caliper=0.25)
```

Similarly to the regional models, the matched data for the three protection types were merged for each country, and GAMM models were fitted for each country’s matched data, using the same formula but with terms relevant to country removed.

Chapter 3

Indigenous lands in protected areas have high forest integrity across the tropics

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3.1. Summary

Intact tropical forests have high conservation value (Watson et al., 2018). Though perceived as wild (Fletcher et al., 2021), they have been under long-term human influence (Roberts et al., 2017). As global area-based conservation targets increase, the ecological contributions of Indigenous peoples through their governance institutions and practices (Dawson et al., 2021) are gaining mainstream interest. Indigenous Lands—covering a quarter of Earth's surface (Garnett et al., 2018) and overlapping with a third of intact forests (Fa et al., 2020)—often have reduced deforestation, degradation, and carbon emissions compared to non-protected areas and protected areas (Sze, Carrasco, et al., 2022; Walker et al., 2020). A key question with implications for the design of more equitable and effective conservation policies is to understand the impacts of Indigenous Lands on forest integrity and long-term use, as critical measures of ecosystem health included within the post-2020 Global Biodiversity Framework (CBD, 2020). Using the Forest Landscape Integrity Index (Grantham et al., 2020) and Anthromes (Ellis et al., 2021) datasets, we find that high-integrity forests tended to be located within the overlap of Protected Areas and Indigenous Lands (Protected-Indigenous Areas). After accounting for location biases through statistical matching and regression, Protected-Indigenous Areas had the highest protective effect on forest integrity and lowest land-use intensity relative to Indigenous Lands, Protected Areas, and non-protected controls pan-tropically. The protective effect of Indigenous lands on forest integrity was lower in Indigenous Lands than in Protected Areas and non-protected areas in the Americas and Asia. The combined positive effects of state legislation and Indigenous presence in Protected-Indigenous Areas may contribute to maintaining tropical forest integrity. Understanding management and governance in Protected-Indigenous Areas can help states appropriately support community-governed lands.

3.2. Results and Discussion

3.2.1. Tropical Indigenous Lands, forest integrity and Anthromes

Intact, high-integrity forests are important for conservation and planetary functioning (Watson et al., 2018), with forest integrity a key measure of ecosystem health in the post-2020 Global Biodiversity Framework (CBD, 2020). Intact forests have been defined as seamless mosaics of a minimum area of 500 km^2 with no remotely detected signs of human activity (Potapov et al., 2017) and bear similarities to high-integrity forests, which conceptually refers to areas with minimal anthropogenic modification to its structure, composition, and function (Grantham et al., 2020). To understand how Indigenous presence and long-term use affects forest integrity across the tropics, we used the Indigenous Peoples' Land dataset (Garnett et al., 2018) and World Database of Protected Areas (UNEP-WCMC & IUCN, 2020) to identify 3.4 Mkm^2 of ILs, 2 Mkm^2 of PAs, 1.7 Mkm^2 of PIAs, and 11 Mkm^2 of non-protected areas (Figure S1).

More than half (56.4%) of tropical forested areas were in the Americas, with 26.8% in Asia and 16.7% in Africa (Figure 1A). The Americas had the highest coverage of PAs and the greatest overlap of PAs and ILs, while Asia had the highest coverage of ILs but lowest coverage of PAs, and Africa had the lowest coverage of PIAs (Table S1A).

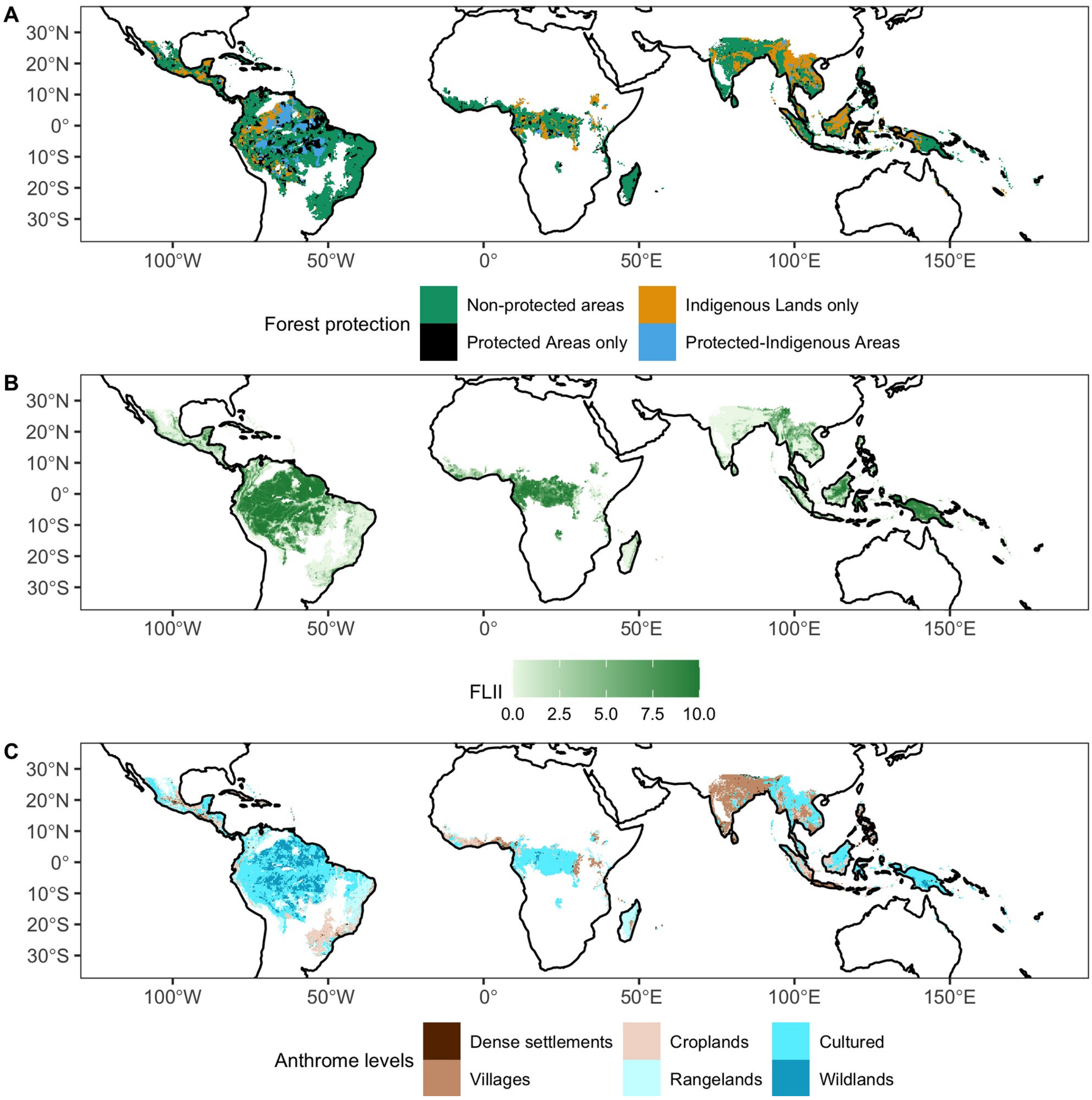

Figure 1. Map of study area across the tropics. A: protection types, intentionally coarsened to 30 km grids with dominant protection type represented to obscure Indigenous land boundaries to prevent inadvertent harm. B: Forest Landscape Integrity Index (FLII) scores, with 10 representing the highest forest integrity score. C: Anthrome levels in 2010. See also Figure S2 and Table S1.

Using the Forest Landscape Integrity Index (FLII) product, which uses satellite-detected disturbances such as road-building, canopy loss, and connectivity loss to model a scaled metric for forest integrity (Grantham et al., 2020), we find that high-integrity tropical forests (where FLII score exceeds 9.6) mirror the distribution of Intact Forest Landscapes (Potapov et al., 2017) with

76.3% overlapping (Figure S2, Table S1B). These high-integrity tropical forests were concentrated in the Amazon and Guiana Shield, Congo, Bornean Highlands, and New Guinea. Lower-integrity tropical forests were prevalent in Central America, the Brazilian Atlantic and Caatinga, West Africa, Indochina, and lowlands of insular Southeast Asia (Figure 1B).

To understand long-term land-use intensity within our study areas, we used the most recent Anthromes data to provide a consistent overview of land-use and intensity over time, characterising landscapes shaped by human interactions with ecosystems (Ellis et al., 2021). The Anthromes data classifies dense settlements, villages, croplands, and rangelands as intensive land-uses; cultured landscapes as low-intensity inhabited areas (with less than 20% intensive-land use); and wildlands as having a complete absence of permanent human populations and intensive land-uses. Most of our study area was covered by low land-use intensity cultured landscapes, such as inhabited drylands and woodlands, in 2010 (Figure 1C), though villages featured prominently in the Indian sub-continent. Only 17.2% of the total study region was considered as wildlands, most of which were in the Amazon. Of the high-integrity forests, only 32.3% were considered as wildlands in 2010 (Figure S2); in Africa and Asia, most high-integrity forests fell under cultured landscapes reinforcing the fact that many areas of conservation importance are not truly wild and human-free but are home to human communities (Fletcher et al., 2021).

While the Americas have high coverage of wildlands, due in part to depopulation following European arrival in 1492, the richness of these forests has also been shaped by pre-colonial forest use (Levis et al., 2017; Roberts et al., 2017). For example, the hyper-diverse Ecuadorian Andean cloud forests were once open fields cultivated by the Indigenous Quijos population (Loughlin et al., 2018). Indigenous communities can enhance forest integrity through management practices that benefit biodiversity (Heckenberger et al., 2007), such as the planting of useful fruit and timber trees and abandonment of plots which result in complex forest structures (Michon et al., 2007). They may also enforce their land rights to keep out infrastructure, (illegal) selective logging, agribusiness expansion, and extractive industries (ICCA Consortium, 2021; Youdelis et al., 2021). On the other hand, they might reduce forest integrity through inadequately regulated timber use or hunting of large-bodied, seed-dispersing vertebrates (Gardner et al., 2019), or be constrained by national infrastructural and economic development plans (Bebbington et al., 2018), exemplified by increasing environmental conflicts (Scheidel et al., 2020).

Most forest integrity measures, including the FLII, rely on remote sensing, which only captures human influences directly detectable by satellites like land-cover changes. This biases them towards monitoring for industrial-scale impacts including mega infrastructure, motorised transport networks, and monoculture plantations, missing other aspects of forest health such as faunal diversity (Plumptre et al., 2021) and forest composition. Nonetheless, the FLII captures other anthropogenic impacts such as hunting and edge effects by modelling inferred pressures, and provides a measure of the degree to which forest structure has been altered.

Comparing FLII scores by protection types within each tropical region (Figure 2A), we found that on average, non-protected areas had the lowest forest integrity (7.21±3.04 (mean±standard deviation)), followed by ILs (7.82±2.61) and PAs (9.04 ±1.96), while Protected-Indigenous Areas (PIAs) had the highest forest integrity (9.48±1.31; Table S2). In the Americas, only non-protected areas had <50% of their area covered by high-integrity forests (Figure 2B). High-integrity forests covered >50% of PAs and PIAs and 44.4% of ILs in Africa, while in Asia, they covered >50% of PIAs, 36.6% of PAs, and only 24.8% of ILs. PIAs thus support a large area of high-integrity tropical forests, though this may be due to biases in locations far from deforestation and forest-use pressures (Joppa & Pfaff, 2009) that could confound their protective effect.

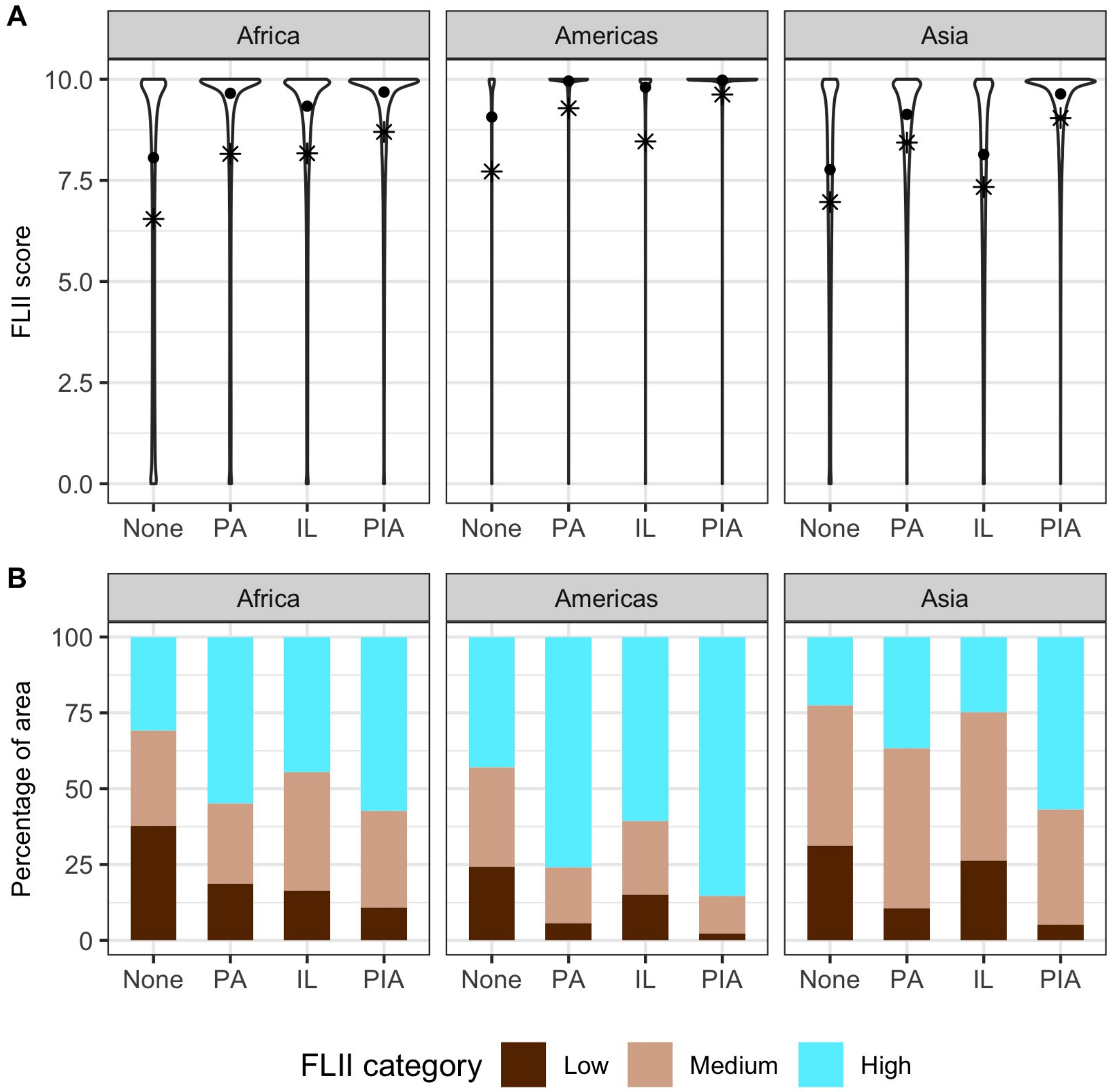

Figure 2. Distribution of FLII in non-protected areas, protected areas only (PA), Indigenous lands only (IL), and protected-Indigenous areas (PIA) across tropical regions. A: violin plots of FLII values for each protection type, with mean value represented by the asterisk symbol and median value represented by the filled circle. B: percentage of forest protection type within FLII categories, where FLII values 0-6=Low, 6-9.6=Medium, 9.6-10=High. See also Table S2.

3.2.2. Effect of protection type on forest integrity

To account for potential confounders in location biases, we used propensity score matching to identify comparable areas of protection types (see STAR Methods). These matched areas covered 444,985 km^2 of ILs, 490,353 km^2 of PAs, 356,745 km^2 of PIAs, and 1,355,865 km^2 of non-protected areas. To predict the effect of protection types on forest integrity within matched areas (see Methods), we ran Generalised Additive Mixed Models. We found mixed results across different tropical regions (Figure 3; Table S1C). In Africa, PAs, ILs, and PIAs all had greater protective effect on forest integrity than non-protected areas by 3-5.2%, with PIAs having the highest protective effect. In the Americas and Asia, PAs and PIAs had greater protective effect than non-protected areas by 1-3%, while ILs had lower protective effect than non-protected areas by 0.1-3.5%. Repeating our analysis with only multi-use PAs (IUCN categories V and VI) found similar results (Figure S3).

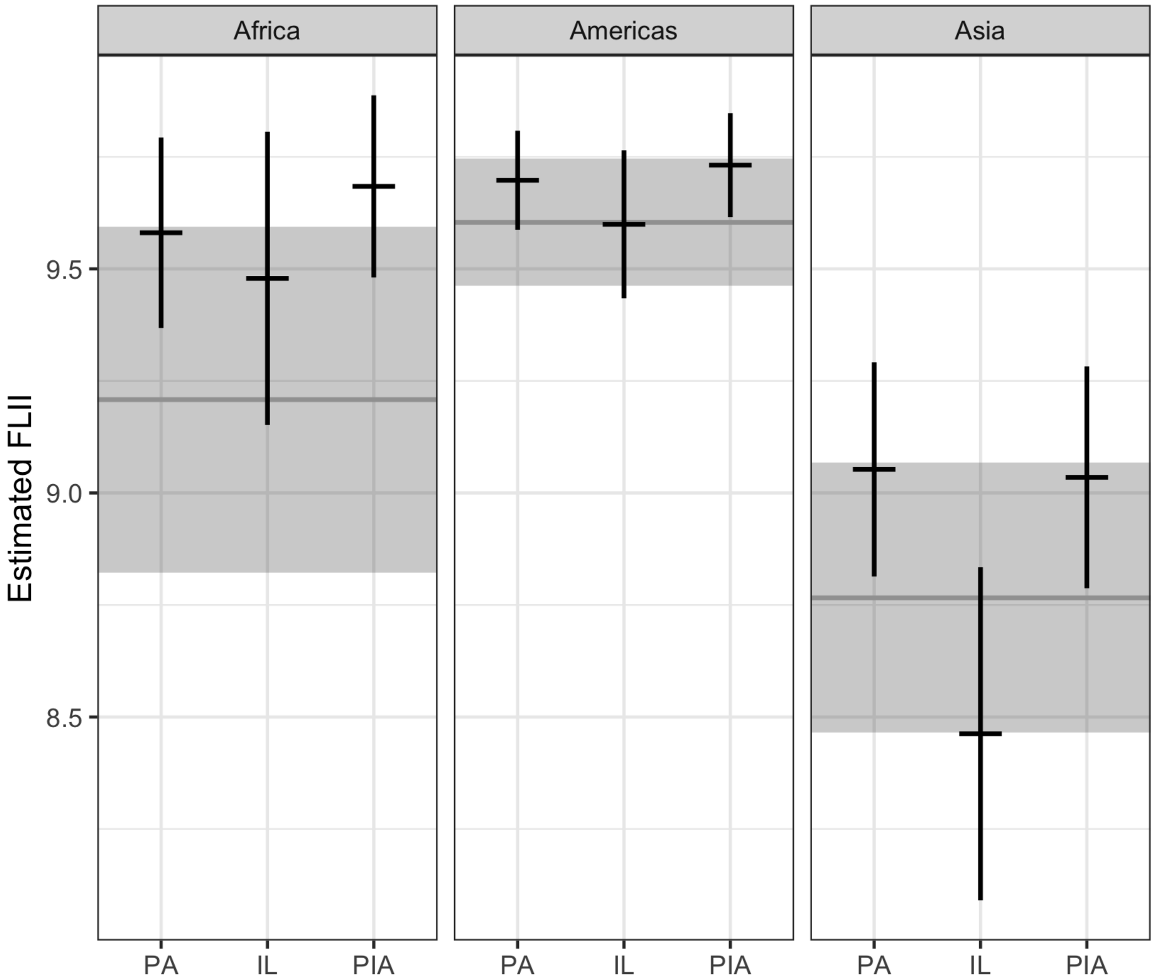

Figure 3. Estimated FLII scores of protection types based on regional GAMMs. Black horizontal lines represent the estimated FLII score, and vertical lines represent their standard errors. Grey horizontal lines represent the estimated FLII score for non-protected areas, and grey shaded area represent its standard error. See also Figure S3 and Table S1.

Forest integrity in ILs in the Americas and Asia scored lower than non-protected areas, while areas that intersect with PAs (i.e., PIAs) scored higher, suggesting that state legislation impacts on forest integrity in these tropical regions. While this may appear to contradict previous work on reduced deforestation and degradation in ILs (Sze, Carrasco, et al., 2022), FLII incorporates inferred deforestation pressures and lost forest connectivity that reflect larger-scale development pressures.

Most ILs, especially in Asia, have not been legally recognised (RRI, 2020), which may partially contribute to their having lower forest integrity than non-protected areas. This is compounded by the fact that the majority of mineral, oil, and gas deposits are located within Indigenous lands worldwide, attracting exploitation by extractive industries and governments for revenue-generation (Anongos et al., 2012). Indeed, governments in countries ranging from Brazil to the Philippines have used the COVID-19 pandemic to pass laws enabling forest exploitation at the expense of Indigenous and local communities (AIPP et al., 2020; Dressler, 2021; Resende et al., 2021).

High economic growth rates in Latin America and Asia have stimulated the expansion of extractive industries in these tropical regions (European Parliament, 2014). That PAs and PIAs retained high forest integrity even after accounting for location biases suggests legal protection hinders forest fragmentation and large-scale developments, corroborating findings that forest PAs mitigate anthropogenic pressures (Geldmann et al., 2019). Across the Amazon, Indigenous territories with legal tenure have reduced deforestation (D. Nepstad et al., 2006; Nolte et al., 2013; Vergara-Asenjo & Potvin, 2014), and mobilised to protect their lands from extractive industries, such as the Munduruku peoples of Brazilian Amazon resisting hydropower development (Walker & Simmons, 2018). However, without state legislation and support, the expansion of extractive industry, infrastructural development, and their associated impacts (which may or may not be desired by Indigenous communities) have impacted tropical forest integrity in ILs and non-protected areas.

Our findings are mediated by the limitation that the datasets we used for ILs and PAs do not indicate the nature of management and/or governance relationship within their areas, nor do they provide information on tenure status, which are critical to producing socially equitable and positive conservation outcomes (Armitage et al., 2020). The data on ILs may also be incomplete, omitting areas where Indigenous peoples are present and influence the landscape while including areas where they are no longer present or have influence. In addition, our study only considered Indigenous lands, which overlooks areas inhabited by local communities who may also have similar positive long-term relationships with their lands (WWF et al., 2021).

3.2.3. Anthromes changes from 1950 to 2010

To further understand how land-use intensity is associated with protection types and has changed across the tropics, we examined land-use intensity using the Anthromes dataset within our matched areas (see Methods). From 1950 to 2010, there has been an increase in land-use intensity across all tropical regions in all protection types (Figure 4; Table S3).

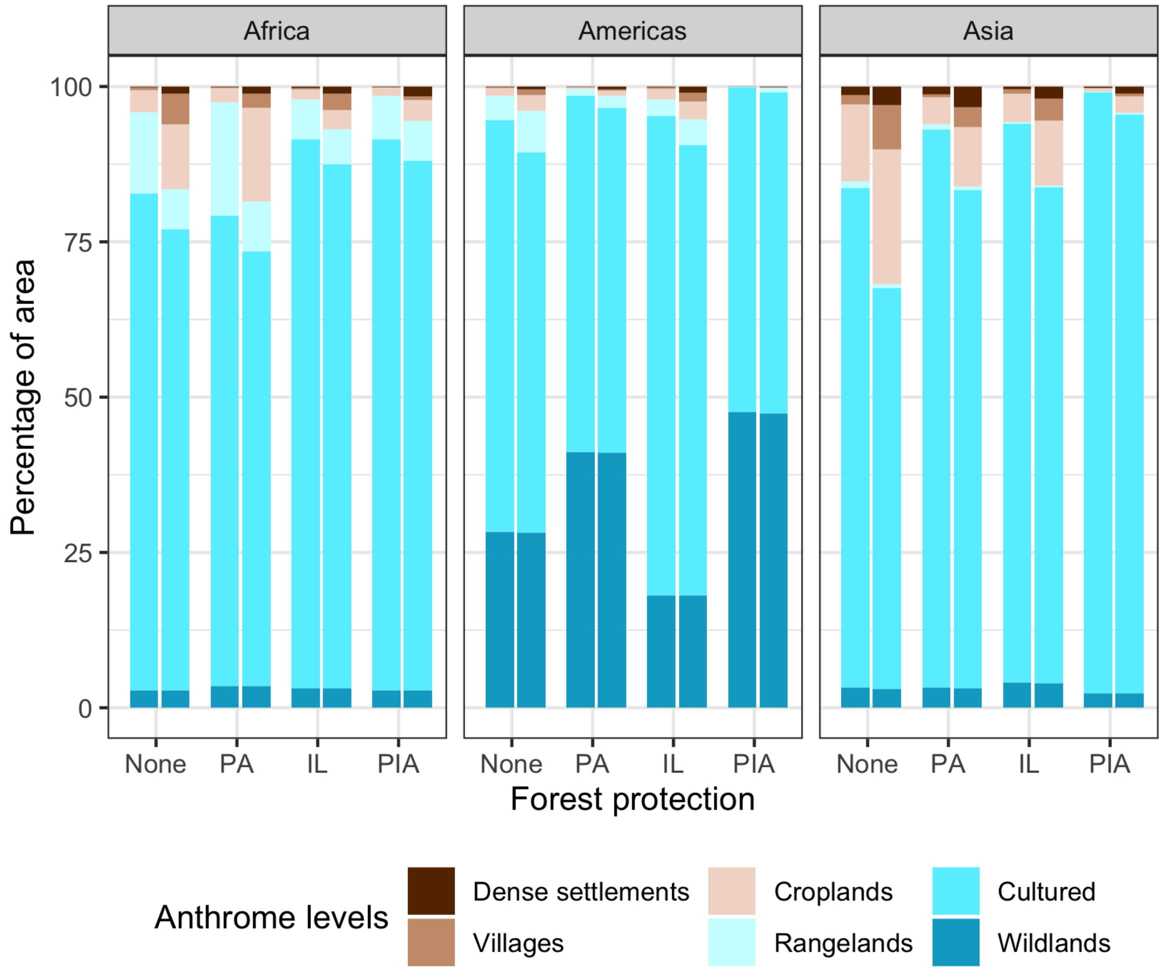

Figure 4. Percentage coverage of Anthrome levels within matched areas. 1950 data is represented in the left column and 2010 data in the right column for each protection type. See also Table S3.

Dense settlements and villages increased six-fold on average, with the largest increase in Africa, followed by the Americas and Asia. Similarly, the largest increase in croplands was in Africa, followed by the Americas and Asia. Most of this expansion towards more intensive land-uses came at the expense of cultured landscapes, and wildlands remained consistent in coverage, reflecting globalised and industrial intensification of land-use (Kastner et al., 2022).

The Americas held the highest coverage of wildlands at 33.7%, while in Africa and Asia, wildlands covered only about 3.1% of the study area. By 2010, intensive land-uses (dense settlements, villages, croplands, and rangelands) covered between 0.9-32.5% of the different protection types. On average, 22% of non-protected areas were intensively used, compared to only 5.8% of PIAs.

Although the Anthromes dataset could overstate the extent to which the Earth has been transformed by human action (Sayre et al., 2017), human influence on landscapes does not necessarily imply degradation, and other maps converge on similar estimates of human influence (Riggio et al., 2020). Our regional-scale spatial analysis found that land-use intensity has increased over time, with PIAs experiencing the least increase; however, local-scale case studies would improve our understanding of its implications and socio-environmental impacts.

3.2.4. Priority of industry-free over human-free

There is increasing clarity on the impact of capitalist-driven extractive industry on areas of conservation concern, including intact forest landscapes (Grantham et al., 2021). The expansion of industrial agriculture into tropical forests has displaced local peoples to more ecologically marginal areas, causing further degradation and deforestation in frontiers (Levers et al., 2021). Additionally, mega-infrastructure and economic growth threaten ecosystem integrity and achieving conservation targets (Johnson et al., 2020; Moranta et al., 2021).

Although we found that PAs can still mitigate large-scale infrastructural development, established PAs across the tropics are being downgraded, downsized, and degazetted to allow for industrial-scale resource extraction (Golden Kroner et al., 2019; Mascia et al., 2014). They can also suffer from inadequate funding, encroachment for hunting, logging or clearance (Coad et al., 2019). Countries worldwide have begun implementing post-COVID economic stimulus plans and policies authorising industrial and extractive activities within PAs (Golden Kroner et al., 2021). These are likely to impact the ability of PAs to continue meeting their conservation objectives.

Conservation policies have moved beyond only area-based targets to encompass metrics including ecosystem integrity and management effectiveness (CBD, 2020). However, the different forms of human-nature relationships and governance arrangements between Indigenous peoples and local communities and state authorities are also important for achieving positive and equitable social and environmental outcomes. Future research could focus on how aspects of livelihoods, biocultural and relational values (Merçon et al., 2019), knowledge systems (Ens et al., 2021), and power dynamics (McDermott et al., 2022) within PAs, ILs, and in particular, PIAs, impact forest integrity and connectivity. This would further our understanding of the contexts in which different conservation policies would be suitable.

3.2.5. Recognising Indigenous Lands in Protected Areas

Our study found the overlap between ILs and PAs to have high forest integrity and minimal intensive and extensive human land-use across the tropics. These spaces often have complex governance relationships, particularly around the recognition and respect of Indigenous peoples' rights (Stevens et al., 2016). In Indonesia, for example, the creation of Betung Kerihun National Park imposed restrictions on the Dayaks contrary to their customary law, creating distrust and resentment (Sunkar & Santosa, 2018), while the Kayan Mentarang National Park is now co-managed by Indigenous authorities, with their customary lands legally recognised (Anau et al., 2019). Yet even where conservation areas are, or appear to be, proposed and managed by Indigenous communities, the relationship often remains tenuous due to histories of coloniality (Rubis & Theriault, 2020; Youdelis et al., 2021) and participation can remain representational (Paulson et al., 2012).

The GBF will likely increase the global land area under some form of protection; safeguards are needed to ensure communities who have not contributed to damaging ecosystems, or may be actively contributing to protecting ecosystem integrity, are not harmed in the process of securing conservation outcomes (Haenssgen et al., 2022; Obura et al., 2021). While our results point to the value of legal protection, the nature and form of legal protection of areas and how they articulate and interact with the Indigenous peoples and local communities already living there will need to be carefully negotiated. Legal recognition for Indigenous peoples and their territories can achieve better socio-ecological outcomes when the process of acquiring legal recognition, the form and extent of legal rights, and the implementation of those rights are navigated sensitively according to their specific contexts (Larson & Springer, 2016).

Pursuing global targets without due attention to power imbalances amongst local governance actors often results in social inequity and failed environmental objectives (McDermott et al., 2022). Equitable ways forward include promoting alternative models to strict PAs in conservation priority areas, such as "territories of life" where communities retain their land ownership and tenure rights and actively govern their lands for conservation and community well-being (ICCA Consortium, 2021). Conservation could also move towards providing funding support through mechanisms such as a Conservation Basic Income to buffer against the economic pressures of industrial extractivism (Fletcher & Büscher, 2020), and co-establishing conservation plans with communities, such as in Pastaza, Ecuador (Selibas, 2022).

3.3. Acknowledgements

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3.4. Author Contributions

Conceptualisation, J.S.S, D.Z.C., L.R.C., and D.P.E.; Data curation and formal analysis, J.S.S.; Writing – initial draft, J.S.S.; Writing – review & editing, J.S.S, D.Z.C., L.R.C., and D.P.E.; Visualisation, J.S.S.; Supervision, D.Z.C., L.R.C., and D.P.E.

3.5. Declaration of Interests

All authors declare no competing interests.

3.6. STAR Methods

3.6.1. Key Resources Table

REAGENT or RESOURCE	SOURCE	IDENTIFIER
Software and algorithms		
R version 4.1.1	The R Foundation	https://www.r-project.org
R Studio version 1.4.1717	RStudio	https://www.rstudio.com/products/rstudio/download/
QGIS version 3.4	Open Source Geospatial Foundation	https://www.qgis.org/en/site/forusers/download.html
ArcMap version 10.7.1	ESRI	http://desktop.arcgis.com/en/arcmap/
Custom code	Author’s own	https://github.com/JocelyneSze/PhD-Ch3-ForestIntegrity
Other		
World Database of Protected Areas (Jan 2020)	UNEP-WCMC and IUCN	https://www.protectedplanet.net/
Indigenous Peoples’ Land	Garnett et al., 2018	Requested from author
Forest Landscape Integrity Index	Grantham et al., 2020	https://www.forestlandscapeintegrity.com/download-data
Anthromes (years 1950 and 2010)	Ellis et al., 2021	https://dataverse.harvard.edu/dataverse/anthromes_12k/

Intact Forest Landscapes	Potapov et al., 2017	https://www.intactforests.org/data.ifl.html
Ecoregions	Dinerstein et al., 2017	http://ecoregions2017.appspot.com/
Spatial Database of Planted Trees	Transparent World and GFW	https://data.globalforestwatch.org/documents/gfw::planted-forests/about
Tree Plantations dataset	GFW	https://data.globalforestwatch.org/datasets/gfw::tree-plantations/explore
Slope	Amatulli et al., 2018	http://www.earthenv.org/topography
Elevation	Amatulli et al., 2018	http://www.earthenv.org/topography
Population density	Lloyd et al., 2017	https://www.worldpop.org/geodata/listing?id=64
Travel time (travel time to nearest urban area with at least 5000 inhabitants in 2015)	Nelson et al., 2019	https://figshare.com/articles/travel_time_to_cities_and_ports_in_the_year_2015/7638134/3
Distance to roads, from SEDAC gRoads v1 with ArcMap Distance toolset to calculate distance from each raster cell to nearest road	Author's own; CIESIN	https://sedac.ciesin.columbia.edu/data/set/groads-global-roads-open-access-v1/data-download
Forest area in 2010, from Global Forest Watch tree cover in 2010 at 25% canopy cover threshold	Author's own; GLAD 2013	https://glad.umd.edu/dataset/global-2010-tree-cover-30-m
Country polygons (GADM version 3.6)	GADM	https://gadm.org/download_world.html

3.6.2. Resource availability

Lead Contact

Further information and requests for information should be directed to and will be fulfilled by the Lead Contact, Jocelyne S. Sze (jssze1@sheffield.ac.uk).

Materials Availability

This study did not generate new unique reagents.

Data and Code Availability

- All datasets used can be downloaded from the original sources or requested from the respective authors as listed in the Key Resources Table.
- All original code has been deposited in the GitHub repository listed in the Key Resources Table and is publicly available as of the date of publication.
- Any additional information required to reanalyse the data reported in this paper is available from the lead contact upon request.

3.6.3. Experimental Model and Subject Details

Study site information

We focused on tropical forest biomes (Tropical and Subtropical Moist Broadleaf Forests, Tropical and Subtropical Dry Broadleaf Forests, and Tropical and Subtropical Coniferous Forests biomes) (Dinerstein et al., 2017), clipping all data layers to their extent and rasterizing layers to 1 km^2 resolution. Spatial data processing was done in R (R Core Team, 2021), QGIS (QGIS, 2020) and ArcMap (ESRI, 2019) in Mollweide equal-area projection (ESRI 54009), before transformation to geographic coordinate system (EPSG 4326) and eventual conversion to a standard data frame. A spatial mask was applied to exclude planted areas from study (Spatial Database of Planted Trees (Global Forest Watch, 2019) and Tree Plantations (Transparent World & Global Forest Watch, 2019)), as well as the 1777 PAs established after our baseline year (i.e., 2010).

3.6.4. Method Details

Forest protection type map

We identified three broad categories of protecting land within tropical moist forests: Indigenous Lands, Protected Areas (PAs), overlapping Protected-Indigenous Areas (PIAs), as well as non-protected areas as controls (Figure S1). Indigenous Peoples' Lands is a database of areas where Indigenous peoples have land tenure or *de facto* management (Garnett et al., 2018). Although areas mapped as Indigenous may not fully be under Indigenous control, and areas not mapped as Indigenous are not necessarily non-Indigenous, we take the data to represent land where Indigenous peoples have likely had influence. PAs, as listed within the World Database on Protected Areas (UNEP-WCMC & IUCN, 2020), are designated by the respective state and legislated to be protected for conservation purposes.

We downloaded protected areas using the R package `wdpar` (Hanson, 2020). We included terrestrial PAs of all categories designated to year 2010, which we took as the baseline year. Following the recommended protocol for cleaning data, we removed non-established sites, UNESCO man and biosphere sites, and point sites due to lack of area information, with additional manual editing in QGIS to remove self-intersections. PAs that had no establishment year (value of 0 in the STATUS_YEAR column) were assumed to have been established before the study period and were included. To obtain PA-only areas, we filtered out PAs that were governed by Indigenous people as listed in the attribute table and removed areas of the PIA spatial intersection, resulting in 3955 PAs.

We obtained spatial data on ILs from the authors and intersected PAs not listed as governed by Indigenous peoples. This spatial intersection was joined with PAs governed by Indigenous peoples (467 PAs) to create Protected-Indigenous Areas (PIAs). To obtain IL-only areas, we removed areas of the PIA spatial intersection from the Indigenous Peoples' Land data.

The remaining areas that fall outside PAs (to January 2020), ILs, or PIAs were considered non-protected (Table S1A). These vector data were then rasterised to 1 km^2 pixels, eliminating double-counting. All pixels that touched the borders of PAs, ILs, or PIAs, were also excluded from the study.

Anthrome levels

We selected Anthrome maps for the years 1950 and 2010 (our baseline year) (Ellis et al., 2021). We include anthromes for 1950 as it represents a time period before extensive land-use changes (i.e., the Green Revolution), to examine how land-use intensity had changed in the decades prior to our study period, though we note that most PAs may not have been designated at that time. Discrete categories are defined based on population densities and intensive land-use cover at regional landscape scales (~100 km^2). In the broadest classification of anthromes, wildlands are characterised by complete absence of human populations and intensive land-uses, cultured anthromes with less than 20% intensive land-use, and intensive anthromes with more than 20% intensive land-use cover. We used the second-level classification: dense settlements (urban areas), villages (dense agricultural settlements), croplands (lands used mainly for annual crops), rangelands (lands used mainly for livestock grazing), cultured (inhabited lands with minor use for permanent agriculture and settlement), and wildlands (lands without human populations or substantial land-use). Dense settlements, villages, croplands, and rangelands are classified as intensive land-uses.

Quantification and Statistical Analysis

Identifying comparable areas with matching

All analyses were conducted in R (R Core Team, 2021). As locations of PAs are biased towards remote, steep, and high-elevation areas (Joppa & Pfaff, 2009), which also affects the likelihood of forest disturbance, we used statistical matching to identify forest areas that would be more comparable. Following Sze, Carrasco, et al. (2022), we included variables that affect forest disturbance and assignment of PAs: slope (Amatulli et al., 2018), elevation (Amatulli et al., 2018), population density (Lloyd et al., 2017), travel time to nearest urban area (Nelson et al., 2019), distance to road (CIESIN, 2013), baseline forest area (GLAD, 2013), and country (Global Administrative Areas, 2018) (Table S4).

We used the MatchIt package (Ho et al., 2018) to conduct propensity score matching for each protection type and unprotected area within each tropical region, following Brooks et al. (2016)'s convention of sorting overseas territories according to geography rather than governing territory. For each of the nine sets of matching, we drew five samples, with replacement, from the full dataset to keep it computationally tractable, resulting in ~85,000 matched pixels for each set. Matching was done with the default logit method, 1:1 nearest neighbour match without replacement, and caliper size of 0.25 to ensure good matches. If no matches were available within the specified calipers, we opted to take the nearest available match. We included all numeric covariates (slope, elevation, population density, travel time, distance to roads, baseline forest area) with country as an exact match, and checked that balance was improved from the matching (Table S5). We then combined the data from across the five matched samples to create a map of matched protection types, representing more comparable areas, covering 444,985 km^2 of ILs, 490,353 km^2 of PAs, 356,745 km^2 of PIAs, and 1,355,865 km^2 of non-protected areas.

Overlay of Anthromes on matched areas

We overlaid the Anthrome layers on our matched areas and counted the number of pixels of each anthromes level within different protection types for the years 1950 and 2010 (Figure 3, Table S3).

Estimating forest integrity

We used the FLII as a measure of forest integrity (Grantham et al., 2020), which improves on the widely-used Intact Forest Landscapes (IFL) (Potapov et al., 2017) by creating a scaled index that additionally incorporates inferred forest pressure by modelling based on proximity to observed pressures to account for edge effects and other human use of forests, such as hunting. We overlaid FLII over our study area to provide an overview of how forest integrity is distributed across the tropics (Figure 1B, Figure 2, Table S2). We also included comparison with the IFL (Figure S2, Table S1B). Observed pressures are mapped from infrastructure (e.g., military, energy generation, and transport infrastructure), agricultural croplands, and recent deforestation, combined with models of inferred pressure and lost forest connectivity to create an index ranging from 0 to 10. This index reflects the degree of anthropogenic modification in 2019, with 10 representing forests with no detectable modification. Following Grantham et al. (2020), in addition to reporting mean and median FLII values, we also categorised FLII into three levels of low (0-6), medium (6-9.6), and high (9.6-10) integrity.

Imbalances remaining in the covariates after matching were accounted for with regression. Split by regions, we had 935930 pixels for Africa, 997714 for Americas, and 532786 for Asia. We fitted generalised additive mixed models for each region using the `mgcv` package (Wood, 2019), including a parametric term for protection type and numeric covariates (slope, elevation, population density, travel time, distance to roads, baseline forest area) as cubic regression smoothing splines. Slope, elevation, travel time, and distance to roads were heavily right-skewed and were cube-root transformed, while population density was transformed by ^(1/5) for Africa and Asia and ^(1/6) for the Americas. We fitted country-level random slopes for 2010 forest area, 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, and quasi-binomial distribution (rescaling FLII to 0 to 1).

To estimate the effect of protection type on FLII, we took median values of numeric covariates and combinations of each country and protection type to create the prediction dataset. We then excluded country effects to isolate the effect of protection types when running the prediction. Additionally, since strict PAs (categories I to IV) often preclude human activities that may affect our results, we repeated our analysis, including only multi-use PAs (categories V and VI) within PA protection type, as a robustness check (Figure S3).

3.7. Supplementary Information

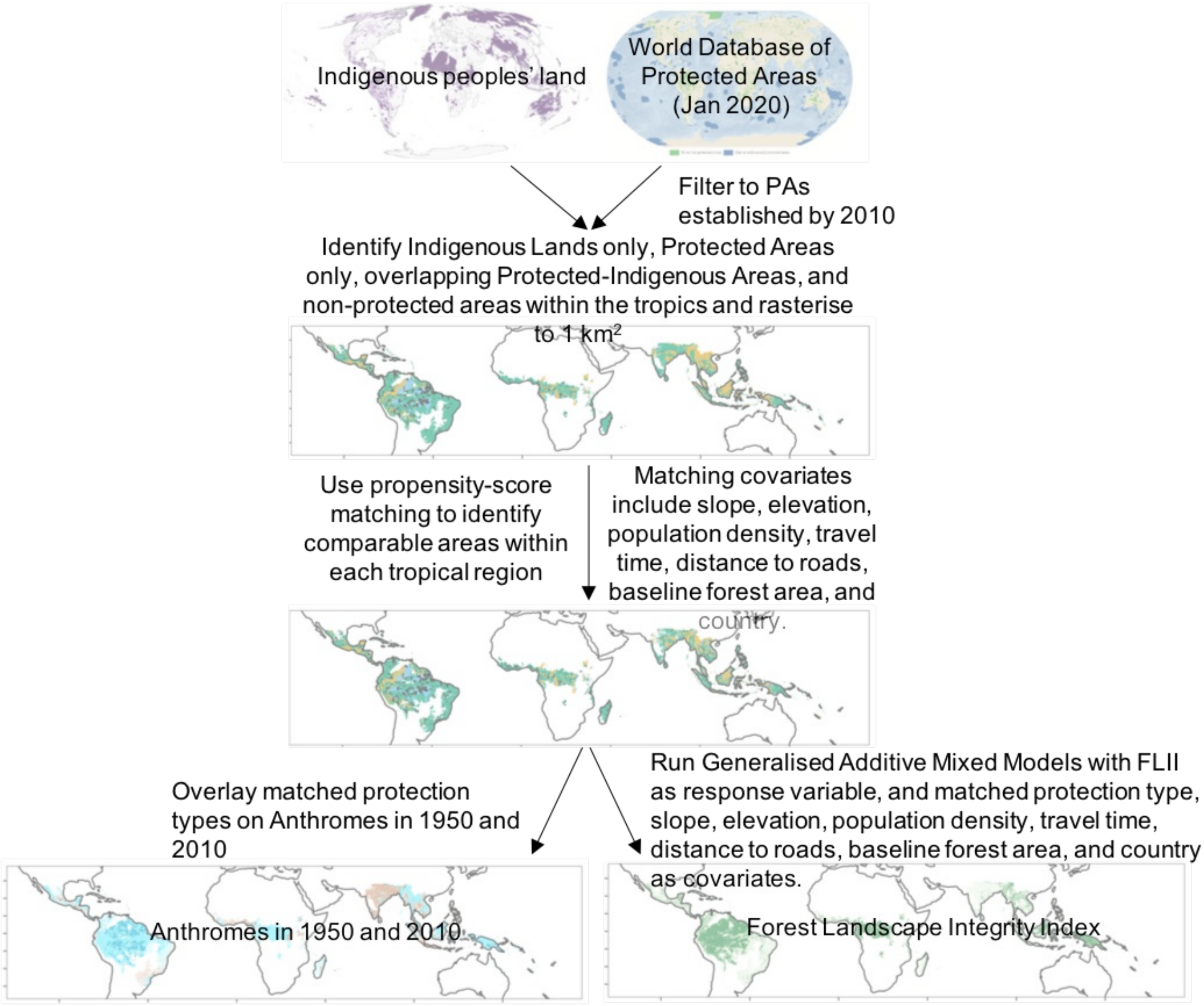


Figure S1. Visual workflow of main datasets used, Related to STAR Methods

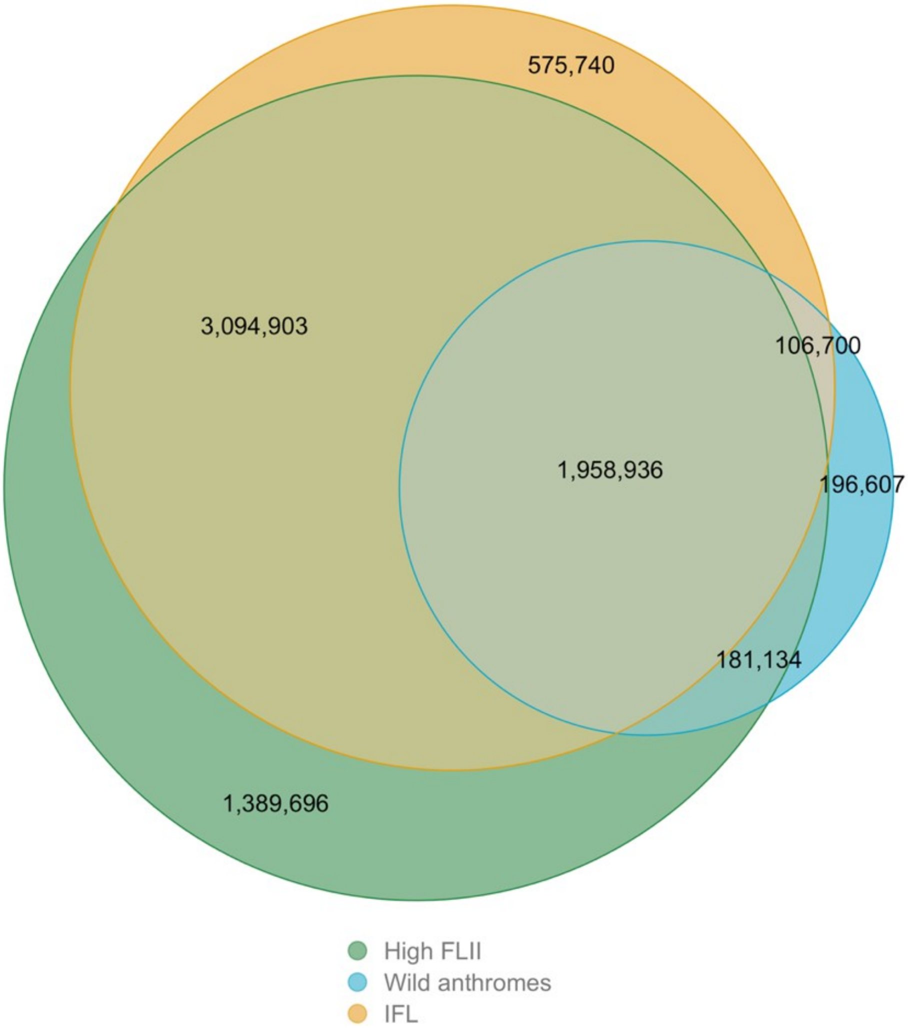


Figure S2. Venn diagram showing areas (in km^2) of high Forest Landscape Integrity Index (FLII) category forests in green, wild anthromes in blue, Intact Forest Landscapes (IFL) in yellow, and their overlaps. Related to Figure 1.

Total area of IFL is 5,736,279 km^2, high-integrity forest is 6,624,669 km^2, and wild anthromes is 2,443,377 km^2. 76.2% of high-integrity forests fall within IFLs (5,053,839 km^2) and 32.3% of high-integrity forests fall within wild anthromes (2,140,070 km^2).

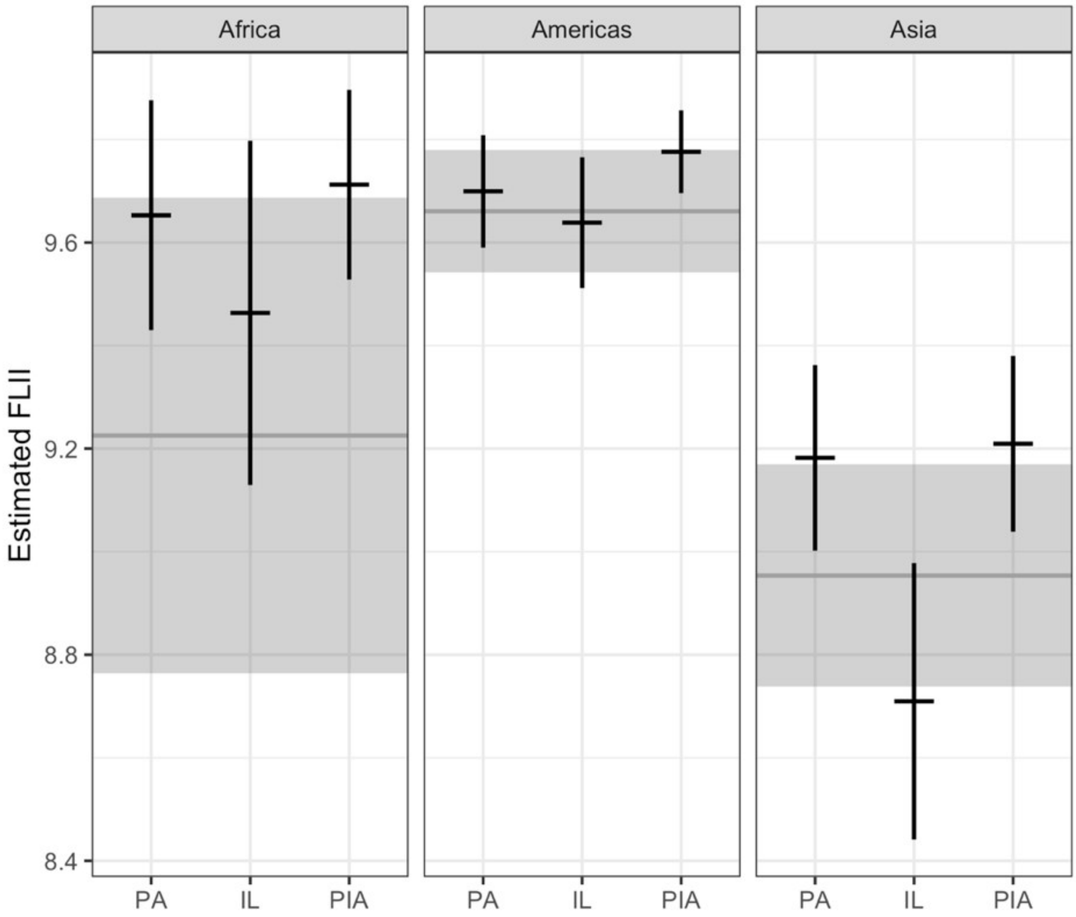


Figure S3. Estimated FLII scores of protection type effects when strict PAs (IUCN categories I-IV) were excluded from the PA protection type. Related to Figure 3. Grey horizontal lines represent value for non-protected areas, grey shaded area represent standard error for non-protected areas and vertical lines represent standard errors for other protection types.

Table S1. Variables of concern in each protection type for each tropical region. Related to Figure 1, STAR Methods, and Figure 3.

A: Area of forest across the tropics in Million km^2, Related to Figure 1.

B: Coverage of Intact Forest Landscapes in percentage, Related to STAR Methods.

C: Estimated FLII scores and standard error, Related to Figure 3.

IPBES regions	**Non-protected areas**	**Protected Areas**	**Indigenous Lands**	**Protected-Indigenous Areas**
A: Area of forests across the tropics in Million km^2 (% of total area in the region)				
Africa	2.16 (71.1)	0.30 (9.9)	0.48 (15.6)	0.10 (3.4)
Americas	6.35 (61.8)	1.54 (15.0)	1.03 (10.0)	1.35 (13.2)
Asia	2.53 (51.8)	0.17 (3.5)	1.88 (38.6)	0.30 (6.1)
B: Coverage of Intact Forest Landscapes in percentage terms				
Africa	19.28	42.18	36.63	44.3
Americas	17.45	59.93	43.51	81.13
Asia	4.6	12.22	8.18	33.23
C: Estimated FLII scores and standard error [percent difference from non-protected area and percent error]				
Africa	9.21±0.38	9.58±0.21 [4.0±4.9%]	9.48±0.33 [2.9±5.6%]	9.68±0.20 [5.2±4.9%]
Americas	9.6±0.14	9.7±0.11 [1.0±1.9%]	9.6±0.17 [-0.1±2.3%]	9.73±0.12 [1.3±1.9%]
Asia	8.77±0.30	9.05±0.24 [3.3±4.5%]	8.46±0.37 [-3.5±5.4%]	9.03±0.25 [3.1±4.5%]

Table S2. Mean Forest Landscape Integrity Index score and breakdown of FLII categories for each protection type and tropical region. Related to Figure 2.

		Non-protected areas	**Protected Areas**	**Indigenous Lands**	**Protected-Indigenous Areas**
Africa	Mean FLII score (s.d.)	6.55 (3.48)	8.16 (2.82)	8.17 (2.52)	8.70 (2.12)
	N pixels	1996192	318730	486838	110472
FLII category (N pixels (%))	Low	754728 (37.8)	59433 (18.7)	80181 (16.5)	12070 (10.9)
	Medium	624716 (31.3)	84529 (26.5)	190435 (39.1)	35201 (31.9)
	High	616748 (30.9)	174768 (54.8)	216222 (44.4)	63201 (57.2)
Americas	Mean FLII score (s.d.)	7.72 (2.77)	9.28 (1.66)	8.46 (2.57)	9.62 (1.14)
	N pixels	3441991	1518897	1005884	1496946
FLII category (N pixels (%))	Low	835001 (24.3)	85000 (5.6)	152880 (15.2)	43049 (2.3)
	Medium	1126596 (32.7)	280285 (18.4)	242881 (24.1)	184414 (12.3)
	High	1480394 (43.0)	1153612 (76.0)	610123 (60.7)	1278483 (85.4)
Asia	Mean FLII score (s.d.)	6.96 (2.81)	8.43 (1.88)	7.33 (2.57)	9.04 (1.48)
	N pixels	1688374	150126	1695540	307835
FLII category (N pixels (%))	Low	527229 (31.2)	15810 (10.6)	447327 (26.4)	15962 (5.2)
	Medium	781121 (46.3)	79333 (52.8)	827312 (48.8)	116784 (37.9)
	High	380024 (22.5)	54983 (36.6)	420901 (24.8)	175089 (56.9)
Entire Tropics	Mean FLII score (s.d.)	7.21 (3.04)	9.04 (1.96)	7.82 (2.61)	9.48 (1.31)
	N pixels	7126557	1987753	3188262	1915253
FLII category (N pixels (%))	Low	2116958 (29.7)	160243 (8.1)	680388 (21.3)	62081 (3.2)
	Medium	2532433 (35.5)	444147 (22.3)	1260628 (39.6)	336399 (17.6)
	High	2477166 (34.8)	1383363 (69.6)	1247246 (39.1)	1516773 (79.2)

Table S3. Classification of Anthrome types in each protection type and tropical region in 1950 and 2010, given as number of cells (percentage within each protection type), Related to Figure 4.

		Intensive Anthromes								**Cultured Anthromes**		**Wild Anthromes**	
		Dense settlements		**Villages**		**Croplands**		**Rangelands**		**Cultured**		**Wildlands**	
		1950	**2010**	**1950**	**2010**	**1950**	**2010**	**1950**	**2010**	**1950**	**2010**	**1950**	**2010**
Africa	None	789 (0.16%)	5777 (1.14%)	2233 (0.44%)	25186 (4.97%)	18207 (3.59%)	52891 (10.4%)	66200 (13.1%)	32930 (6.5%)	405005 (79.9%)	375697 (74.1%)	14392 (2.84%)	14345 (2.83%)
	PA	161 (0.1%)	1761 (1.1%)	252 (0.16%)	3873 (2.41%)	3687 (2.3%)	24022 (15.0%)	29359 (18.3%)	12963 (8.08%)	121539 (75.7%)	112385 (70.0%)	5527 (3.44%)	5521 (3.44%)
	IL	424 (0.25%)	2068 (1.21%)	438 (0.26%)	4419 (2.59%)	2585 (1.52%)	5198 (3.05%)	11107 (6.51%)	9624 (5.64%)	150641 (88.3%)	143886 (84.3%)	5401 (3.17%)	5401 (3.17%)
	PIA	92 (0.09%)	1574 (1.61%)	5 (0.01%)	598 (0.61%)	1424 (1.45%)	3292 (3.36%)	6839 (6.98%)	6276 (6.41%)	86889 (88.7%)	83509 (85.2%)	2734 (2.79%)	2734 (2.79%)
Americas	None	554 (0.12%)	2153 (0.45%)	567 (0.12%)	4125 (0.87%)	5860 (1.24%)	12503 (2.64%)	18888 (3.99%)	31581 (6.66%)	313818 (66.2%)	289827 (61.2%)	134167 (28.3%)	133665 (28.2%)
	PA	281 (0.14%)	1014 (0.49%)	78 (0.04%)	406 (0.2%)	422 (0.2%)	1769 (0.86%)	2307 (1.12%)	4024 (1.95%)	118596 (57.4%)	114548 (55.4%)	85011 (41.1%)	84934 (41.1%)
	IL	215 (0.14%)	1546 (0.99%)	274 (0.18%)	2185 (1.41%)	2790 (1.8%)	4579 (2.95%)	4078 (2.62%)	6325 (4.07%)	119983 (77.2%)	112721 (72.6%)	28037 (18.0%)	28021 (18.0%)
	PIA	43 (0.03%)	208 (0.13%)	3 (0.0%)	35 (0.02%)	25 (0.02%)	207 (0.13%)	186 (0.11%)	1052 (0.65%)	84599 (52.3%)	83640 (51.7%)	76932 (47.6%)	76646 (47.4%)

Asia	None	3276 (1.38%)	7118 (2.99%)	3440 (1.45%)	17058 (7.17%)	29762 (12.5%)	51613 (21.7%)	2404 (1.01%)	1412 (0.59%)	191255 (80.4%)	153389 (64.5%)	7684 (3.23%)	7231 (3.04%)
	PA	1221 (1.22%)	3323 (3.31%)	533 (0.53%)	3216 (3.2%)	4333 (4.31%)	9687 (9.64%)	813 (0.81%)	472 (0.47%)	90332 (89.9%)	80646 (80.3%)	3234 (3.22%)	3122 (3.11%)
	IL	370 (0.47%)	1533 (1.95%)	582 (0.74%)	2800 (3.57%)	3532 (4.5%)	8204 (10.5%)	234 (0.3%)	169 (0.22%)	70528 (89.9%)	62626 (79.8%)	3199 (4.08%)	3113 (3.97%)
	PIA	211 (0.18%)	1366 (1.18%)	25 (0.02%)	574 (0.49%)	650 (0.56%)	3035 (2.62%)	244 (0.21%)	228 (0.2%)	112238 (96.7%)	108166 (93.2%)	2686 (2.31%)	2685 (2.31%)
Across the tropics	None	4619 (0.38%)	15048 (1.36%)	6240 (0.51%)	46369 (7.1%)	53829 (4.42%)	117007 (13.8%)	87492 (7.18%)	65923 (6.23%)	910078 (74.7%)	818913 (60.4%)	156243 (12.8%)	155241 (11.1%)
	PA	1663 (0.36%)	6098 (1.62%)	863 (0.18%)	7495 (3.04%)	8442 (1.81%)	35478 (9.5%)	32479 (6.94%)	17459 (6.65%)	330467 (70.7%)	307579 (61.0%)	93772 (20.0%)	93577 (18.2%)
	IL	1009 (0.25%)	5147 (1.68%)	1294 (0.32%)	9404 (5.38%)	8907 (2.2%)	17981 (7.72%)	15419 (3.81%)	16118 (4.97%)	341152 (84.4%)	319233 (72.4%)	36637 (9.06%)	36535 (7.86%)
	PIA	346 (0.09%)	3148 (0.92%)	33 (0.01%)	1207 (0.74%)	2099 (0.56%)	6534 (2.4%)	7269 (1.93%)	7556 (2.84%)	283726 (75.5%)	275315 (72.1%)	82352 (21.9%)	82065 (21.0%)

Table S4. Data name, description, manipulation done and source. Related to STAR Methods

Variable	Description	Processing done	Source
Forest Landscape Integrity Index (`FLII`)	Index of forest integrity based on observed pressures, inferred pressures and lost forest connectivity (min 0, max 10)	Original pixel size ~300 m. Aggregated by factor of 3 taking the mean value, resampled to ~1 km pixel using 'near'	Grantham et al., 2020 https://www.forestlandscapeintegrity.com/download-data
Intact Forest Landscape (`IFL`)	Map of contiguous forests (at least 500 km^2) with no remotely detected signs of human activity in 2013	Rasterised vector data to ~1 km pixel	Potapov et al., 2017 https://www.intactforests.org/data.ifl.html
Anthromes (`anthromes1950AD`, `anthromes2010AD`)	Anthrome classification for years 1950 and 2010.	Original pixel size ~ 10 km. Resampled to ~1 km pixel using 'near'	Ellis et al., 2021 https://dataverse.harvard.edu/dataverse/anthromes_12k/
Protection Type (`Type`)	Type of forest protection: None, PAs, ILs, or PIAs	See description below	IUCN and UNEP-WCMC, 2020; Garnett et al., 2018 https://www.protectedplanet.net/en
Slope (`slope`)	Mean slope (degrees) in ~1 km pixel	Original pixel size ~1 km. Resampled to harmonise dataset using 'bilinear'	Amatulli et al., 2018 http://www.earthenv.org/topography
Elevation (`elevation`)	Mean elevation (metres) in ~1 km pixel	Original pixel size ~1 km. Resampled to harmonise dataset using 'bilinear'	Amatulli et al., 2018 http://www.earthenv.org/topography
Population Density (`ppp`)	Mean number of people in a ~1 km pixel in 2010	Original pixel size ~1 km. Resampled to harmonise dataset using 'bilinear'	Lloyd et al., 2017 https://www.worldpop.org/geodata/listing?id=64
Travel Time (`travelTimeto5kcity`)	Travel time to nearest urban area with at least 5000 inhabitants in 2015 (mins)	Original pixel size ~1 km. Resampled to harmonise dataset using 'bilinear'	Nelson et al., 2019 https://figshare.com/articles/travel_time_to_cities_and_ports_in_the_year_2015/7638134/3
Distance To Roads (`DisttoRoads`)	Euclidean distance to nearest road (metres)	We clipped the road geospatial layer to tropical forest extent, re-projected to EPRI 54009, and rasterised it. Used the `distance` tool, `Euclidean	CIESIN, 2014 https://sedac.ciesin.columbia.edu/data/set/groads-global-roads-open-

		distance planar` in ArcMap to calculate distance from each raster cell to the nearest road raster cell, and re-projected back to EPSG 4326.	access-v1/data-download
Forest Area (`Forest2010Area`)	Area in ~1 km pixel that was forested in 2010 at 25% tree canopy threshold (m^2)	We multiplied the number of 30 m pixels in each 1 km cell that were forested in 2010 by the mean 30 m-pixel area using the area function in the raster package to obtain baseline forest area.	GLAD, 2013 https://glad.umd.edu/dataset/global-2010-tree-cover-30-m
Country (`gadm36`)	Country boundaries	Rasterised vector data to ~1 km pixel	GADM, 2018 https://gadm.org
Ecoregion	Tropical and subtropical moist broadleaf, dry broadleaf, and coniferous forest biomes	Extracted the three tropical forest biomes for defining study area boundaries	Dinerstein et al., 2017 http://ecoregions2017.appspot.com/

Table S5. Number of pixels (including control and treatment) and mean value of standardised difference in means (SMD) across the six covariates before and after matching with propensity score at the regional level for PAs established before 2011 at tropical moist forest extents for the five samples. Balanced is considered to have improved if Standardised Difference of Means (SMD) is reduced. Related to STAR Methods.

		Africa			**Americas**			**Asia**		
		PA	**IL**	**PIA**	**PA**	**IL**	**PIA**	**PA**	**IL**	**PIA**
Complete Dataset	N pixels	2920015	3121726	2688449	9394688	8802762	9162886	3236908	5302259	3386464
	mean SMD	0.3905	0.2778	0.4483	0.4439	0.3559	1.2761	0.4795	0.5893	1.934
Sample 1	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.3883	0.2786	0.4478	0.4383	0.3594	1.2528	0.468	0.6601	2.0261
After matching	N pixels	95096	91402	93678	105996	86336	73942	95184	56382	71238
	mean SMD	0.0651	0.0768	0.1106	0.0764	0.2907	0.1612	0.1468	0.2443	0.2732
Sample 2	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.4013	0.2776	0.4445	0.4389	0.3535	1.4228	0.4646	0.5656	1.885
After matching	N pixels	95972	90726	94468	105894	85678	73158	95232	56304	70552
	mean SMD	0.0659	0.0775	0.1112	0.0758	0.2884	0.1601	0.1463	0.2357	0.2752
Sample 3	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.3994	0.2814	0.4471	0.4385	0.3618	1.324	0.5009	0.6151	1.8664
After matching	N pixels	95926	90994	94432	105844	85414	73416	95696	57504	71206
	mean SMD	0.071	0.0767	0.1108	0.0728	0.296	0.1622	0.1489	0.2409	0.2708

Sample 4	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.4246	0.2768	0.4413	0.4451	0.35	1.2751	0.5361	0.613	2.048
After matching	N pixels	95356	91330	94468	105722	85732	73534	94864	56500	71448
	mean SMD	0.0709	0.0731	0.1104	0.0751	0.2934	0.169	0.1516	0.2423	0.2818
Sample 5	N pixels	408802	280955	1102264	281840	352110	274886	809227	106045	474104
	mean SMD	0.416	0.2751	0.4582	0.4515	0.3531	1.3648	0.476	0.5875	1.8777
After matching	N pixels	95380	90984	93738	105756	85318	73152	95388	57084	71612
	mean SMD	0.0705	0.0766	0.1114	0.0776	0.2946	0.156	0.1456	0.2376	0.259

Chapter 4

Indigenous Peoples' Lands are critical for safeguarding vertebrate diversity across the tropics

4.1. Abstract

Indigenous Peoples are long-term custodians of their lands, but only recently have their contributions to conservation started to be recognised in biodiversity policy and practice. Tropical forest loss and degradation are lower in Indigenous lands than unprotected areas, yet the role of Indigenous Peoples' Lands (IPLs) in biodiversity conservation has not been properly assessed from regional to global scales. Using species distribution ranges of 11,872 tropical forest-dependent vertebrates to create Area of Habitat maps, we identified the overlap of these species ranges with IPLs and then compared values inside and outside of IPLs for species richness, extinction vulnerability, and range-size rarity. Of assessed vertebrates, at least 76.8% had range overlaps with IPLs, on average overlapping ~ 25% of their ranges; at least 120 species were found only within IPLs. Species richness within IPLs was highest in South America, while IPLs in Southeast Asia had highest extinction vulnerability, and IPLs in Dominica and New Caledonia were important for range-size rarity. Most countries in the Americas had higher species richness within IPLs than outside, whereas most countries in Asia had lower extinction vulnerability scores inside IPLs and more countries in Africa and Asia had slightly higher range-size rarity in IPLs. Our findings suggest that IPLs provide critical support for tropical forest-dependent vertebrates, highlighting the need for greater inclusion of Indigenous Peoples in conservation target-setting and programme implementation, and stronger upholding of Indigenous Peoples' rights in conservation policy.

4.2. Introduction

The Kunming-Montreal Global Biodiversity Framework aims to increase the land and sea area under protection to 30% by 2030 (CBD, 2022), including Indigenous and traditional territories. The recognition of the integrity and distinct nature of Indigenous and traditional territories to area-based conservation has significance for addressing the historical and ongoing social harms caused by conservation (e.g. land dispossession, violence, and intergenerational trauma) and recognising the consistency of many Indigenous environmental practices with the Convention on Biological Diversity goals (Reyes-García et al., 2021; Tauli-Corpuz et al., 2020).

Indigenous Peoples' Lands (IPLs)—areas which are owned, managed, and/or used by Indigenous Peoples—cover more than a quarter of the Earths' terrestrial surface (Garnett et al., 2018). These areas overlap with 36% of Intact Forest Landscapes (Fa et al., 2020) and are increasingly recognised as critical for global biodiversity maintenance (Brondízio et al., 2021; Reyes-García et al., 2021). At least 25% of tropical forests fall within IPLs, where they have reduced deforestation and degradation relative to non-protected areas (non-PAs) and perform similarly to protected areas (PAs) (Nolte et al., 2013; Sze, Carrasco, et al., 2022). Possibly arising from a combination of legal protection and Indigenous management, areas of IPLs that overlap PAs also have higher forest integrity than non-PAs, suggesting they confer more ecosystem services such as carbon sequestration, water and climate regulation, while fulfilling important material and non-material needs for Indigenous and local communities and providing habitat for biodiversity (Grantham et al., 2020; Sze, Childs, et al., 2022).

While the case for the global significance of Indigenous stewardship has been developed by Indigenous leaders, scholars, and philosophers for decades, if not longer (e.g., Atleo, 2012; Salmón, 2000), geospatial analysis is increasingly used to quantify the contributions of Indigenous Peoples to global biodiversity conservation. For example, the ranges of many mammal species overlap with IPLs, with 2695 (of 4460) species having ≥10% and 1009 species >50% of their ranges on these lands (O'Bryan et al., 2020). In addition, 47% of threatened mammals occur on IPLs, and for more than a quarter of them, >50% of their ranges are on these lands (O'Bryan et al., 2020). IPLs overlap the ranges of 71% of the world's 521 primate species (Estrada et al., 2022) and support more vertebrate species than existing PAs or randomly selected non-PAs in Australia, Brazil, and Canada (Schuster et al., 2019; see also Corrigan et al., 2018).

Although the relevance of IPLs for mammals is apparent, particularly at local to regional scales (Fernández-Llamazares et al., 2021; Renwick et al., 2017), our understanding of the coverage of IPLs for other vertebrate taxa at the pantropical level is still very limited. A key question therefore is the degree to which IPLs overlap terrestrial tropical vertebrate biodiversity – spanning amphibians, birds, reptiles as well as mammals – and where the high values of tropical biodiversity are within IPLs globally. Identifying this overlap is key to understanding the various tensions and synergies between current Indigenous environmental practices and global biodiversity conservation efforts underway following the Kunming-Montreal Global Biodiversity Framework.

Focusing on forest-dependent vertebrates of the biodiverse tropics, we examine the importance of IPLs for terrestrial biodiversity by tackling three objectives: (1) Identify the extent to which forest-dependent vertebrate Area of Habitat (AOH) in 2020 overlaps with IPLs, PAs, and non-PAs across the tropics; (2) Identify countries where IPLs contain more species, species at higher risk of extinction, and greater range-size rarity; and (3) quantify whether IPLs contain more species, species at higher risk of extinction, and greater range-size rarity than buffer zones outside of IPLs.

4.3. Methods

4.3.1. AOH for terrestrial tropical forest-dependent vertebrates in 2020

We focused on terrestrial forest-dependent vertebrate groups (amphibians, birds, mammals, and reptiles) that had distributional range maps. Spatial polygons on amphibian, mammal, and reptile distributions were obtained from the IUCN Red List of Threatened Species (IUCN, 2020), and bird distributions from BirdLife International (BirdLife International, 2020). These maps represent known or inferred areas where species occur based on georeferenced observations and expert knowledge. Following Tracewski et al. (2016), we only considered terrestrial species that were native or re-introduced, and extant or possibly extant. We filtered for species that had any part of their range overlapping tropical forests, and whose only preferred habitat was listed as forest for amphibians, mammals, and reptiles, or as having medium or high forest dependency for birds, based on habitat information in the IUCN Red List and BirdLife International, respectively. This yielded 11,872 tropical forest-dependent vertebrates. Tropical forest-dependent vertebrates may also be considered as those that are found exclusively within tropical forests, in addition to forest habitat preferences or dependencies. We thus conducted an additional filter for vertebrates whose distributional ranges were entirely within tropical forest extents; this yielded 1251 vertebrates (Supplementary Material).

Range distribution maps in equal-area Mollweide projection (ESRI: 54009) were then re-projected to geographic latitude/longitude coordinate system (EPSG: 4326) and rasterised to 1 km^2 pixels to obtain each species' Area of Habitat (AOH) (Brooks et al., 2019). Reductions in AOH contribute to heightened species extinction risk (Durán et al., 2020). To obtain the AOH for each species, we cropped its distribution range to forest cover in 2020 (at 50% canopy cover threshold for each 1 km^2 pixel; Hansen et al., 2013) and its altitudinal range when available. Species altitudinal limits were obtained from the IUCN Red List and BirdLife International; for amphibians, where altitudinal limits were provided, a 300 m altitudinal buffer on both upper and lower bounds was added, following Ficetola et al. (2014). We used the Global Forest Change dataset for forest cover in 2020 as it covers a larger extent of the tropics compared to the Tropical Moist Forest dataset (Vancutsem et al. 2022).

4.3.2. *Species' ranges within Indigenous Peoples' Lands, protected areas, and non-protected areas*

Based on their AOH for 2020, 271 of the 11,872 forest-dependent vertebrates had no remaining suitable tropical habitat left, with 96% of these having small ranges to begin with (i.e., area of distribution <20,000 km^2, which may meet the IUCN Red List Criteria B on geographic range to be listed as Vulnerable). Many of these 271 species were located on small island countries such as Samoa, French Polynesia, Micronesia, and Seychelles, which are not covered by satellite-derived tree cover data (Hansen et al. 2013). For the remaining 11,601 species, we identified species that had some of their range overlapping with mapped IPLs (Garnett et al., 2018), PAs (UNEP-WCMC & IUCN, 2020), neither, or both, and calculated the degree of overlap. Areas where IPLs (~15.5 million pixels) and PAs (~6.6 million pixels) overlapped (~2.6 million pixels) were labelled as Protected-Indigenous Areas (PIAs). We identified species that were found only in IPLs exclusive of the overlapping PIAs to understand the additional contribution of IPLs, but kept the overlapping areas for the rest of the analysis, i.e. IPLs and PAs do not specify mutually exclusive areas and both include the overlapping PIAs. Following O'Bryan et al. (2020), for species that overlapped with IPLs, we classified how much of their habitat overlapped at <20%, 20-40%, 40-60%, 60-80%, 80-99%, and 100% levels.

We used the boundaries of Indigenous Peoples' Lands mapped in Garnett et al. (2018), who identified Indigenous lands across 87 countries or politically distinct areas. This dataset represents the most comprehensive assessment of terrestrial lands where Indigenous Peoples have customary ownership, management, and/or governance arrangements in place, regardless of legal recognition.

We acknowledge that voids in these maps do not necessarily imply an absence of Indigenous Peoples or their lands, but rather, areas for which an Indigenous connection could not be determined from publicly available geospatial resources. The definition of Indigeneity adopted in this dataset aligns with those of the International Labour Organisation Indigenous and Tribal Peoples Convention 1989 (No. 169) Article 1 (ILO, 1989).

4.3.3. Countries where IPLs have high values of species richness, extinction vulnerability, and range-size rarity

To identify areas that harbour high numbers of species, highly threatened species, and species with small range sizes, we produced maps for species richness, extinction vulnerability, and range-size rarity. For species richness, we stacked the species' AOH maps to obtain the total number of species occurring in a given pixel, for all vertebrates and for each taxon separately.

For extinction vulnerability, we calculated an extinction vulnerability score for each pixel by taking the mean value of the threat score for all species occurring in a given pixel:

$$Extinction\,vulnerability = \frac{\sum_{i=1}^{n} T_i}{n}$$

where n = number of species occurring in the given pixel and T = the threat score for the species.

We assigned the following numerical threat scores to each IUCN threat category in a geometric progression, following Wang et al. (2020): Least Concern = 2, Near threatened = 4, Endangered = 8, Vulnerable = 16, and Critically Endangered = 32. For the 1004 species that were Data Deficient, we obtained predicted threat categories from Bland et al. (2015), Butchart & Bird (2010), González-del-Pliego et al. (2019), and Jetz & Freckleton (2015). For the remaining 626 species that had no predicted threat category, we calculated the global mean threat score for its taxon, rounded to the nearest integer (i.e., amphibians = 11, birds = 4, mammals = 8, and reptiles = 6). Higher values thus represent pixels containing species in higher threat categories, more vulnerable to extinction.

Range-size rarity highlights areas important for small-ranged species. We calculated range-size rarity as the mean value of the inverse of the AOH for all species occurring in a given pixel:

$$Range = \frac{\sum_{i=1}^{n} \frac{1}{AOH_i}}{n}$$

where n = number of species in the given pixel and AOH = the number of AOH pixels for the species.

We used the inverse of the species' total AOH (Guerin & Lowe, 2015), instead of calculating the proportion of each species' range within a given pixel, since our maps are at the 1 km^2 resolution rather than 10 or 100 km^2. Higher values thus represent pixels that are more important for species with small ranges.

4.3.4. Biological values inside and outside Indigenous Peoples' Lands

To compare whether conservation metrics (species richness, extinction vulnerability, and range-size rarity) have higher values inside or outside IPLs, we created 10 km radius buffer zones around the IPLs. We used 10 km as it is commonly chosen by researchers comparing the effectiveness of PAs with their buffer zones (Fuller et al., 2019). PAs gazetted up to January 2020 were removed from these maps. We opted not to conduct spatial matching to identify counterfactuals for IPLs as it is highly improbable to identify (and obtain the data for) all variables contributing to species diversity patterns that might confound with IPL location, which goes beyond accessibility or remoteness of IPLs. Further, our intention was not to account for location biases (though we acknowledge that IPLs and species are not randomly located) to make comparisons of the effectiveness of IPLs in conserving species diversity given similar baseline conditions. Rather, we wanted to examine species diversity as could currently be found within IPLs and provided a contrast with their 10-km buffer zones as a comparison for understanding what difference (if any) IPLs might make given the varied geographic and historical factors influencing species diversity distribution are likely to be similar between IPLs and their 10-km buffers.

We constructed country-level permutation tests on the difference between the mean value of each conservation metric – species richness, extinction vulnerability, and range-size rarity – among IPLs and 10 km buffer zones. Null distributions were constructed using 1000 permutations at the pixel level, from which p-values were calculated using a two-tailed test (See Supplementary Materials). This analysis was carried out at the country level (53 countries) to account for possible confounders such as national legislation concerning biodiversity, but note that this does not account for the diversity of ecoregions within each country. IPLs cover about 27% of our study area in the tropics, but range between 0.6% to 86.6% for each of the 53 countries where IPLs were mapped (STable 1). We present our results following the United Nations geoscheme for geographic regions, covering Africa (n=15 countries), the Americas (n=21), Asia (n=15), and Oceania (n=2).

We conducted permutation tests for each of the three metrics – species richness, extinction vulnerability, and range-size rarity. We further plot the biophysical attributes (i.e., slope, elevation, population density, and travel time to nearest city of 5000 inhabitants) associated with IPLs and the 10 km buffer zone to understand how such attributes may affect biodiversity. We repeated this analysis using a 50 km radius and all areas outside of IPLs for robustness checks.

4.4. Results

4.4.1. *Species' ranges within Indigenous Peoples' Lands, protected areas, and non-protected areas*

Of the 11,601 forest-dependent vertebrate species, 1456 were amphibians, 6398 birds, 1725 mammals, and 2022 reptiles. Of these species, 70% were Least Concern (n=7107 species) or Near Threatened (977), 9% were Data Deficient (1004), and the remaining 21% were considered threatened (Total = 2513; Vulnerable = 1003, Endangered = 1026, Critically Endangered = 484). Distributional ranges of 8874 vertebrate species (76.5%) intersected with IPLs, encompassing at least 48% of amphibians, 87% of birds, 76% of mammals, and 63% of reptiles.

Nearly all (94.5% or n = 10,965 species) forest-dependent vertebrates had some of their AOH falling within IPLs or PAs, but for 56.6% of them (6205), half or more of their range was outside IPLs and PAs (Figure 1A). The mean average value of range overlap in IPLs was 23.1% for amphibians, 28.7% for birds, 30.1% for mammals, and 26.7% for reptiles, compared to 51.8%, 57.1%, 53.3%, and 54.1% overlap in areas outside of IPLs and PAs for the four taxon groups, respectively. The mean range overlap in PAs was similar to IPLs for reptiles at 26%, higher for amphibians at 33.5%, and lower for birds and mammals at 22.4% and 25.9%, respectively. These range overlap values for both IPLs and PAs include where they overlap each other (Protected-Indigenous Areas), which were 8.4% for amphibians, 8.2% for birds, 9.3% for mammals, and 6.8% for reptiles. Similarly, for threatened species (Figure 1B), the mean range overlap in IPLs (14.7-28.3% for the four taxa) was less than that in areas outside IPLs and PAs (48.1-58.8% for the four taxa). However, the mean range overlap in PAs (24.9-37.2% for the four taxa) was greater than in IPLs for amphibians, birds, and reptiles.

Following Hanson et al. (2020), we considered a variable value of coverage of species' AOH as 'sufficiently protected', ranging from 100% for AOH < 1000 km2 to 10% for AOH > 250,000 km2 with the coverage value log-linearly interpolated for intermediate AOH, and a maximum of 1,000,000 km2 where coverage value > 10,000,000 km2. Given this variable coverage value, 31.3%

of the assessed species (n = 3633 species) would be considered sufficiently protected by PAs alone, and 42.5% (n=4935) by IPLs alone. This increases to 54.8% (n=6361) when both PAs and IPLs are considered. A total of 2728 species (42.9% of the 6361 species) would not be considered sufficiently protected without the additional coverage provided by IPLs. Although we use the terms “sufficiently protected” here as a heuristic, we do not imply that IPLs automatically confer protection on species. We provide this analysis merely as an illustration of the potential substantial contribution that IPLs would provide rto species’ habitat, especially if Indigenous peoples autonomously chose to partake in national conservation planning and were supported and included in relevant processes..

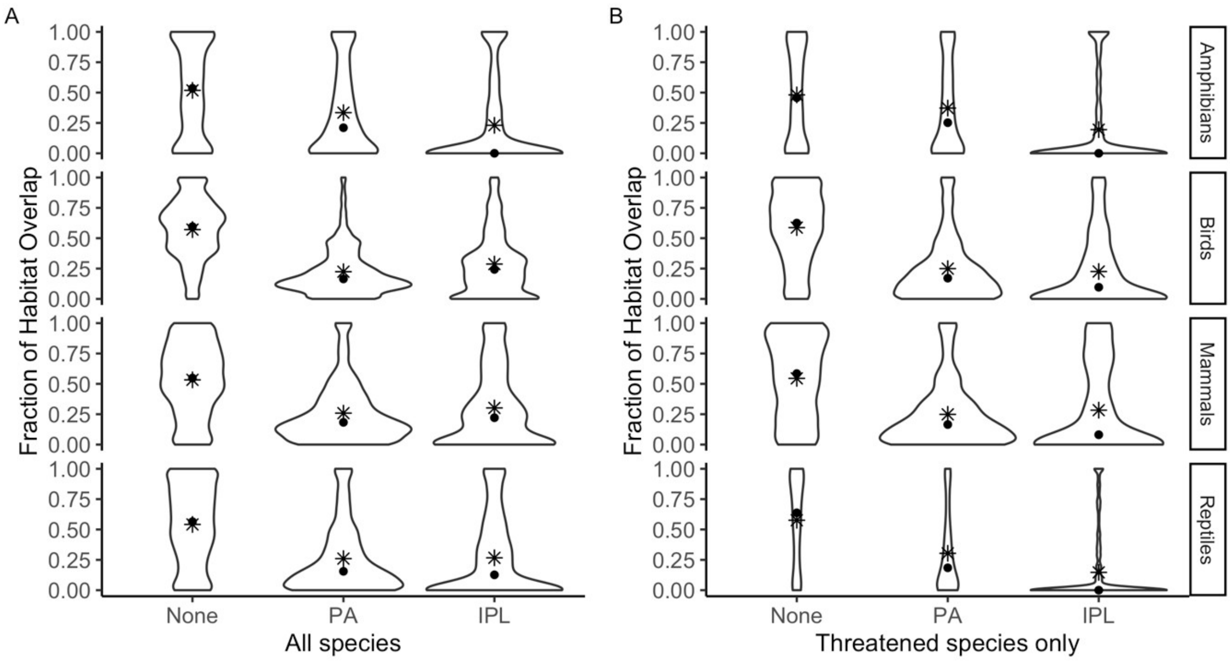


Figure 1. Fraction of Area of Habitat overlap for A. all amphibians, birds, mammals, and reptiles, and B. threatened amphibians, birds, mammals, and reptiles, with areas outside both Indigenous Peoples’ Lands and Protected Areas (None), Protected Areas (PA), and Indigenous Peoples’ Land (IPL). Mean values represented by the asterisk symbol and median values represented by the filled circle. Note that values for PAs and IPLs include where they overlap each other.

Although 51.8-62.4% of species within each vertebrate taxon had half or more of their range outside IPLs and PAs, 120 species were found only within IPLs (exclusive of overlap with PAs), 148 species only within PAs (exclusive of overlap with IPLs), and 52 species only within the overlapping areas of IPLs and PAs (Figure 2A). Of the 120 species found only within IPLs, 53 were amphibians, 6 birds, 18 mammals, and 43 reptiles, and nearly half (n=57) of the 120 species were listed as threatened.

Focusing on the 8874 species whose ranges intersected with IPLs (including areas overlapping PAs), while 729 were classified as Near Threatened and 6221 as Least Concern, 35.4% of the 700 amphibian species, 10.4% of 5567 birds, 23.8% of 1321 mammals, and 12% of 1286 reptiles were threatened (Figure 2B). About 20% (n=1823) of overlapping species had >60% of their range within IPLs (261 amphibians, 855 birds, 347 mammals, and 360 reptiles), of which 42.5% of amphibians, 13.6% of birds, 38.3% of mammals, and 15% of reptiles were threatened. For 288 species, all of their remaining areas of habitat range fell within IPL, encompassing 124 amphibians, 24 birds, 47 mammals, and 93 reptiles. About half of these species in each vertebrate class were threatened, with the exception of reptiles, where most were classified as Data Deficient. Many Data Deficient species are, however, likely to be threatened with extinction (Borgelt et al., 2022).

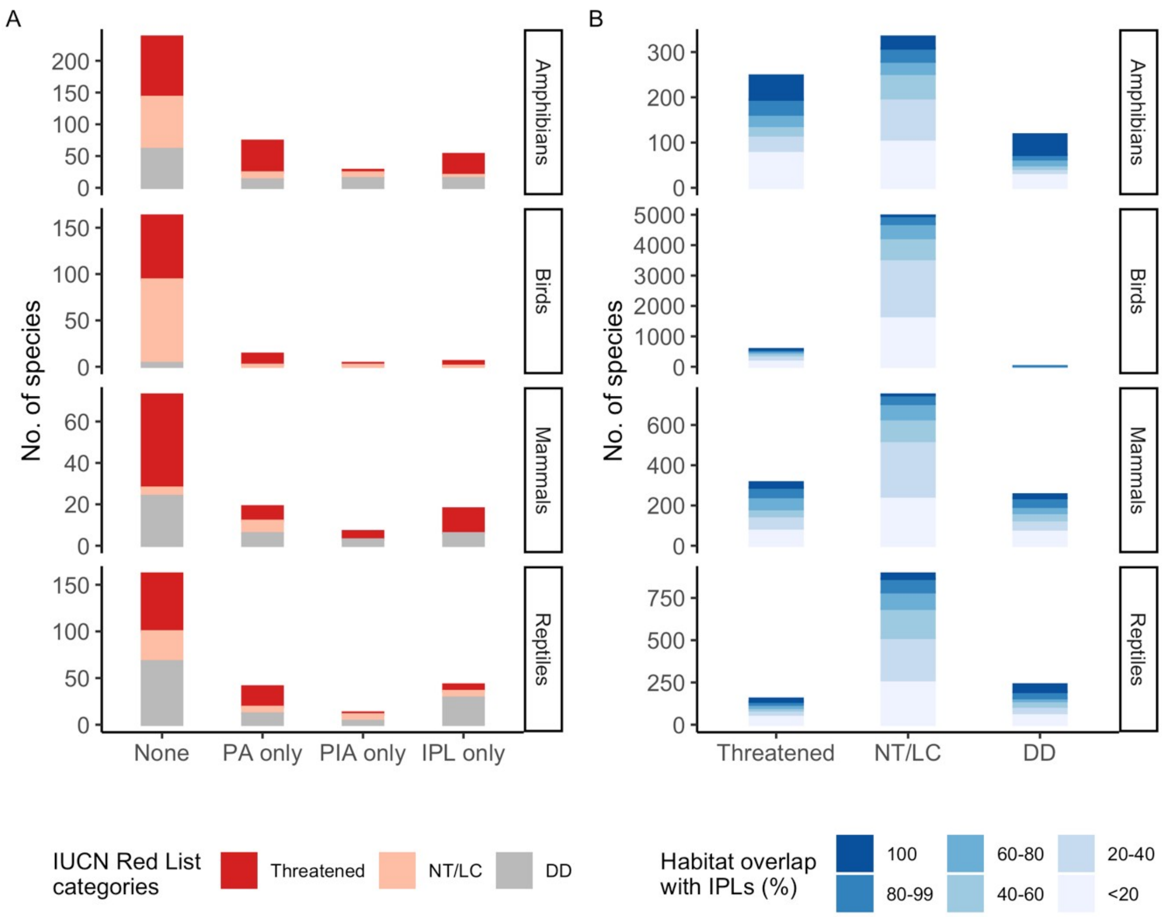


Figure 2. A. Number of species of Data Deficient (DD), Threatened (Critically Endangered, Endangered, and Vulnerable) and Near Threatened/Least Concern (NT/LC) amphibians, birds, mammals, and reptiles (from top to bottom) completely outside of both Indigenous Peoples' Lands and Protected Areas (None), with

all of their range within Protected Areas (PA only), with all of their range within the overlap of Protected Areas and Indigenous Peoples' Lands (PIA only), and with all of their range within Indigenous Peoples' Lands (IPL only). B. Number of species of amphibians, birds, mammals, and reptiles (from top to bottom) with varying % of their range in Indigenous Peoples' Lands which are Data Deficient (DD), Threatened (Critically Endangered, Endangered, and Vulnerable), or Near Threatened/Least Concern (NT/LC).

4.4.2. Countries where IPLs have high values of species richness, extinction vulnerability, and range-size rarity

Across the tropics, median species richness for IPLs was highest in South America (Figure 3A), particularly Ecuador and Peru. Within each region, IPLs with the highest median species richness were in Gabon and the Republic of Congo for Africa, and Malaysia and Laos for Asia.

IPLs with species more vulnerable to extinction were concentrated in Southeast Asia, in particular Malaysia and Indonesia (Figure 3B). In Africa, extinction vulnerability was high in Togo and Benin, and in the Americas, it was Costa Rica and Paraguay. IPLs with higher values of range size rarity—areas important for small-ranged species—were concentrated in the small island nations of Dominica and New Caledonia (Figure 3C; though not clearly visible at the pan-tropical scale). In Africa, Rwanda and Tanzania had the highest range-size rarity, while in Asia, Pakistan and the Philippines had the highest range-size rarity. We also plot the three metric values together to illustrate how each country scored relative to other metrics (SFigure 1).

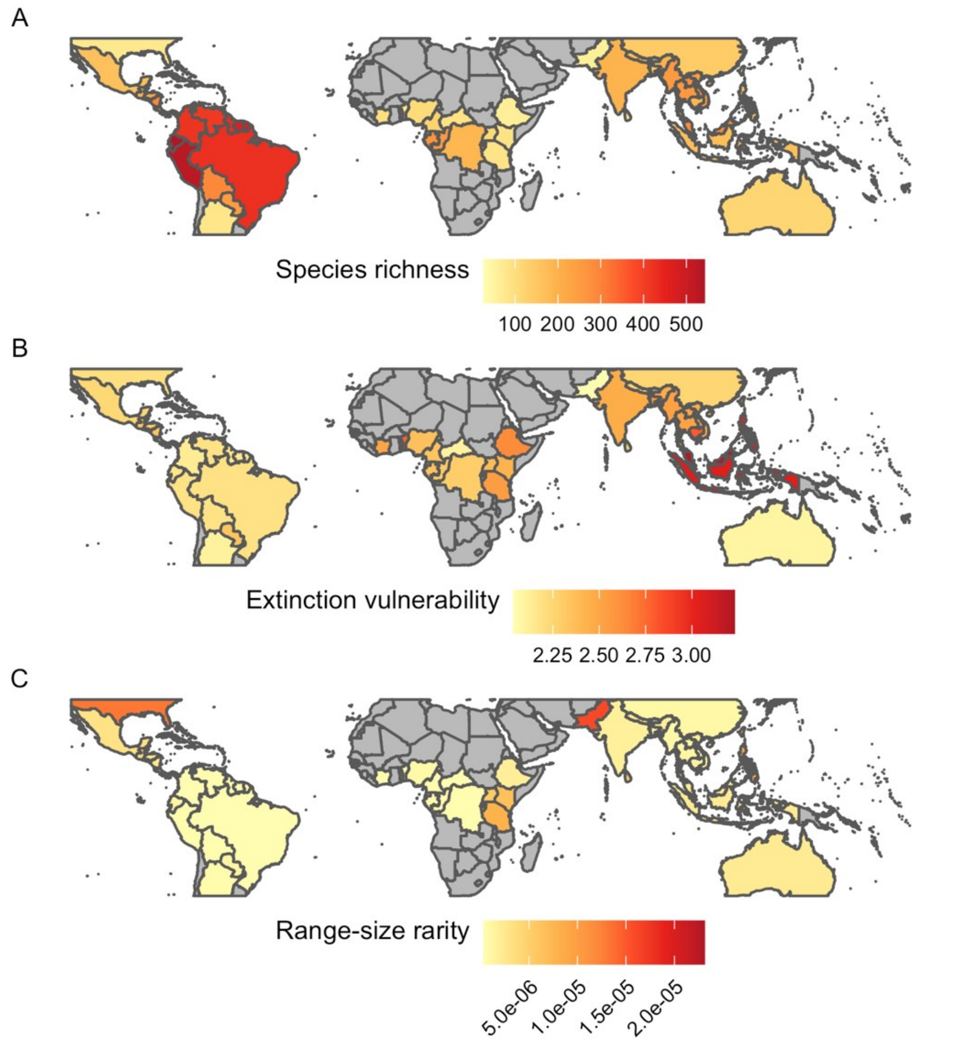


Figure 3. Median values of A. species richness, B. extinction vulnerability, and C. range-size rarity in Indigenous Peoples' Lands within tropical and subtropical forest biomes.

4.4.3. Biological values inside and outside Indigenous Peoples' Lands

Of the 53 tropical countries with IPLs, 27 had significantly higher forest-dependent vertebrate species richness in IPLs than in the 10 km buffer zone outside, while 26 had significantly lower species richness (Figure 4). Countries with higher species richness in IPLs were mostly in the Americas and Asia, while those with lower species richness were mostly in Africa and Oceania. These trends were similar for 50 km buffer zone areas and for all areas outside IPLs (SFigure 2).

Twenty-one countries had more species vulnerable to extinction inside IPLs than in the 10 km buffer zone outside, while 31 had lower extinction vulnerability inside IPLs. Countries with higher species' extinction vulnerability were mostly in the Americas and Africa whereas Asia and Oceania countries mostly had lower extinction vulnerability risk inside IPLs. Differences in range-size rarity inside and outside IPLs were small, except for Dominica and USA (not plotted) which had much lower range-size rarity values inside IPLs than in the 10 km buffer zone. Twenty-one countries had significantly higher values inside IPLs than outside and 28 countries had lower range-size rarity inside IPLs. Most countries in Asia had higher range-size rarity values in IPLs, while those in the Americas and Africa had mostly lower range-size rarity values in IPLs.

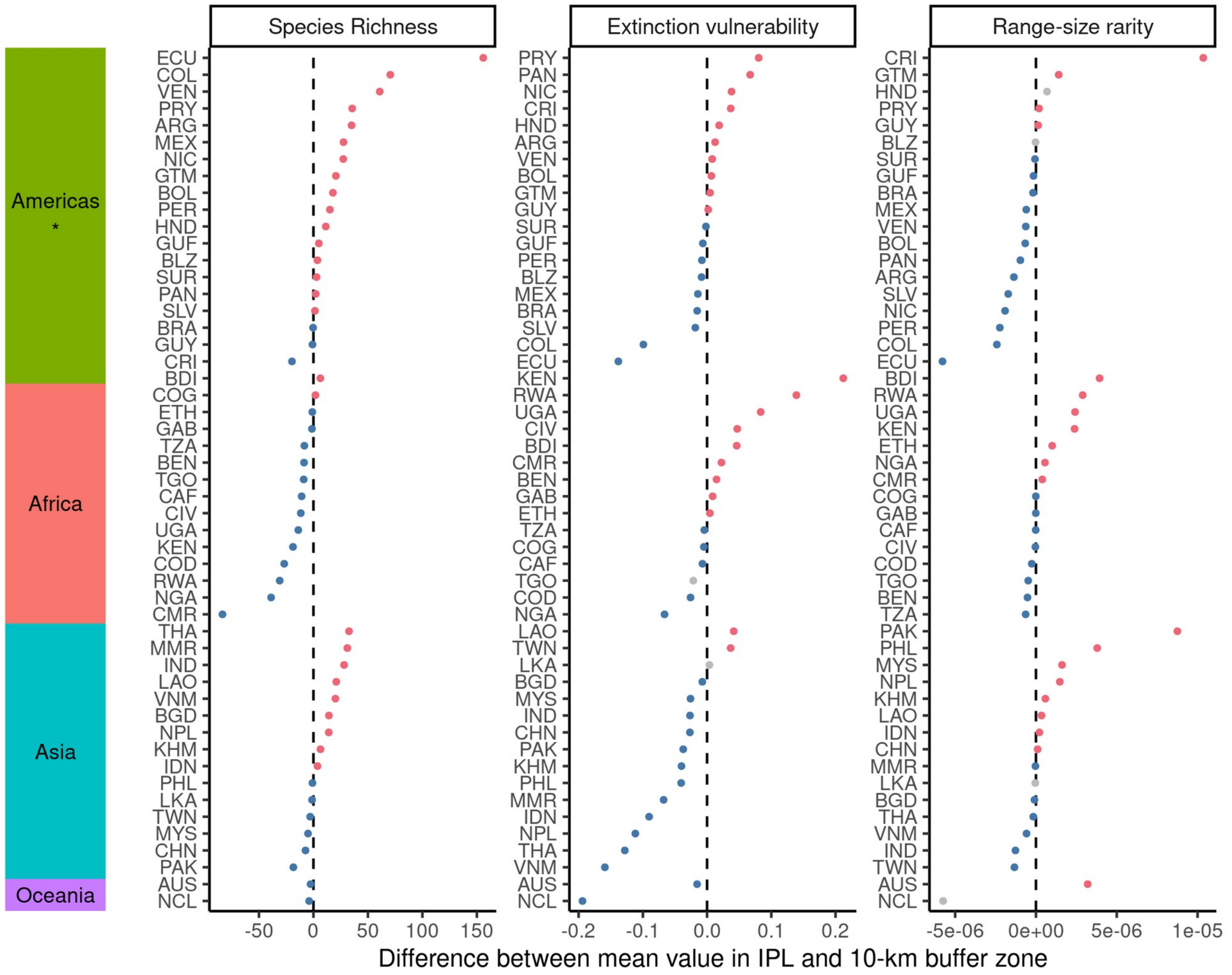


*Figure 4. Difference between mean values of species richness, extinction vulnerability, and range-size rarity within IPL and the 10 km buffer area outside. Red dots represent significantly positive difference (greater value inside IPL), blue dots represent significantly negative difference (smaller value inside IPL), and grey dots represent no significant difference, at alpha=0.05 level. *DMA and USA are not plotted here for visual purposes as outliers; DMA values for species richness=-1.77(sig.neg.), extinction risk=-0.113(sig.neg.), range-size rarity=-0.00005(not.sig.); USA values for species richness=50.9(sig.pos.), extinction risk=-1.88(sig.neg.), range-size rarity=-0.000334(sig.neg.).*

Most of these differences between the mean value in IPL and 10 km buffer zone for species richness were driven by birds—as the most speciose group—as well as mammals and reptiles (SFigure 3). For some countries where overall species richness was significantly lower inside IPLs than outside, such as French Guiana, Costa Rica, Uganda, Rwanda, Philippines, Taiwan, Malaysia, and Australia, amphibian species richness was actually greater inside IPLs. Ecuador also had a much higher amphibian species richness inside IPLs than outside, compared to all the other countries.

For extinction vulnerability (SFigure 4), most of the overall pattern was driven by birds and mammals. Although the overall extinction vulnerability score in IPLs was lower than the 10 km buffer zone for almost all Asian countries, except for Laos and Taiwan, at the taxon level, Malaysia, India, Cambodia, Philippines, Indonesia, and Thailand had higher amphibian extinction vulnerability scores inside IPLs. India and Nepal also had higher mammalian scores inside IPLs, and Malaysia, Indonesia, Nepal, Thailand, and Vietnam had higher reptilian scores inside IPLs.

Most of the overall pattern for range-size rarity was also driven by birds (SFigure 5). However, while for most taxa differences in values inside IPLs and in the 10-km buffer zone were small, for reptiles in Dominica and mammals in Burundi, Rwanda, Uganda, Kenya, Philippines, Nepal, and Australia, IPLs had higher range-size rarity values, meaning they were important for smaller-ranged animals.

We examined how biophysical variables that act as proxies for land-use frequency (slope, elevation, population density, and travel time) were associated with IPLs and the 10 km buffer zone (SFigure 6). We found that the mean difference inside and outside IPLs for elevation and slope were scattered around 0. For population density, mean differences were generally 0 or negative, indicating similar or lower average population densities in IPLs. For travel time, countries in the Americas had mean difference values ranging from -3485 to 5043, while countries in Africa and Asia were generally 0 or negative.

Overall, species richness was highest in IPLs in countries in the Americas, where it was significantly higher than in the 10 km buffer zone. Extinction vulnerability of species in IPLs was highest in countries in Asia, but this tended to still be significantly less than in the 10 km buffer zone. However, half of American and African countries had species with significantly higher extinction vulnerability in IPLs than outside. Range-size rarity of species in IPLs was highest for small island nations, with Costa Rica and Pakistan showing significantly higher range-size rarity inside IPLs.

4.5. Discussion

4.5.1. Forest-dependent vertebrates' AOH within Indigenous Peoples' Lands

Although IPLs cover 28.2% of remaining tropical forest, they provide habitat for about 75% of the vertebrate diversity we assessed. These species may be benefiting from the relatively undisturbed forest habitats retained within IPLs (Estrada et al., 2022). Botanical, archaeological and ethnoecological research have shown that Indigenous communities have shaped tropical forests' structure and composition over millennia through their cultural practices (Levis et al., 2017; Maezumi et al., 2018). It is plausible that these practices might have increased landscape heterogeneity and created highly suitable habitats for many vertebrate species (Fernández-Llamazares et al., 2021). For 23.5% of tropical forest-dependent vertebrates, IPLs provide additional habitat to PAs that would mean a variable coverage of their AOH fall within IPLs or PAs. For example, Giant muntjac (*Muntiacus vuquangensis*) from the Annamite mountain ranges of Laos, Vietnam, and Cambodia has about 24% of its range within PAs, but 72% lies within IPLs. While habitat loss and local hunting have contributed to its population decline (Pin et al., 2022), conservation efforts with local communities may reverse the trend.

For 288 species, IPLs are critical for their survival, containing their entire range. Ixtlan deer mouse (*Habromys ixtlani*) and small-ranged salamanders *Pseudoeurycea saltator*, *P. smithi*, and *Thorius arboreus* are all endemic to the Sierra de Juarez range of Oaxaca, Mexico. Although few designated state PAs exist in this cloud forest, the Indigenous Zapotec and Chinantec communities have come together to manage their forests collectively, and operate a reportedly successful example of community forestry (Chapela, 2005). Such cases demonstrate the existing contributions of Indigenous Peoples and their interwoven knowledge systems and cultural practices to biodiversity conservation efforts. However, this can be contingent on whether lands marked as Indigenous are truly within Indigenous Peoples' ownership, management, and autonomy. Rhaegal's false garden lizard (*Pseudocalotes rhaegal*), endemic to the Cameron Highlands, Malaysia, for example, has its entire global range within IPLs. However, Cameron Highlands is a well-known tourist destination and the Indigenous Semai population has limited decision-making influence and power around the development and management of the land (Ismail et al., 2021), with the species' habitat threatened by expanding agriculture and urban settlements (How Jin Aik et al., 2021). In contrast, reduced forest loss in Oaxaca, Mexico, is attributed to the autonomous Indigenous municipalities that retain meaningful influence on local institutions, compared to neighbouring municipalities with Indigenous areas but without capacity (Haines, 2021).

Even where territories are recognised as under Indigenous ownership or management and protected legally, this often does not prevent exploitation of their lands by external actors (Quijano Vallejos et al., 2020). For example, Santa Marta wren (*Troglodytes monticola*), which is endemic to the Sierra Nevada de Santa Marta in Colombia, is subjected to many industrial development and extractive pressures, such as mining, illicit crops, and unsustainable agricultural intensification. Despite technically being co-managed with the Tayrona Indigenous Confederation representing the Indigenous Kogi, Wiwa, Arhuaco and Kankuamo communities, and demands from them for the area to be free from mining, there has been a lack of enforcement on environmental protection policies (Duran-Izquierdo & Olivero-Verbel, 2021). Having autonomy over land management, supportive policies, and enforcement of these policies is often critical for enabling biodiversity conservation in IPLs. This is particularly pertinent as many countries around the world ramp up their renewable energy transition efforts to mitigate climate change, requiring energy transition minerals and metals that are located on or near lands of Indigenous and peasant peoples (Owen et al., 2022).

4.5.2. IPLs' importance for forest-dependent vertebrates

Despite IPLs covering a relatively small fraction of most species' ranges, they still have high species richness, particularly in the Americas, with 17 countries harbouring significantly more species inside than outside IPLs (excluding PAs). Ecuador, for example, has much higher species richness inside IPLs across all four vertebrate taxon groups. Since our study uses remaining suitable habitat, our finding is likely to be related to higher forest cover retained in IPLs, as was discussed in Fernández-Llamazares et al. (2021) for Amazonian bats. Nonetheless, almost all countries in Africa (except Burundi and the Republic of Congo) had lower species richness inside IPLs than outside, with Cameroon, in particular, having on average 83.5 species fewer. Overhunting, expansion of logging roads, modern technologies, and influx of farmers have contributed to biodiversity declines, leading to the creation of Community Hunting Zones and PAs by the Cameroonian government, both of which have comparable species richness (Bobo et al., 2014). Since we did not use biodiversity field surveys that might more reliably inform species' presence, and the IPL dataset is not likely to correspond with these Community Hunting Zones, further research on the reasons for a much lower species richness in IPLs in Cameroon is needed. Our assessment underscores that a substantial proportion of IPLs in Africa face the threat of conversion as extractive industries expand their area of influence (see also Estrada et al., 2022).

Extinction vulnerability of species within tropical IPLs was highest in Asia, where species were more likely to be critically endangered than they were in other geographic regions. However, when compared to areas outside IPLs, we did not find that species in IPLs were more likely to be endangered, except in Laos and Taiwan. Overall, threats to forest-dependent vertebrates are high in Asia since tropical forests have undergone large-scale conversion to rubber or oil palm monoculture plantations (Warren-Thomas et al., 2015; Wilcove & Koh, 2010). Although most IPLs in Asia lack official recognition and land tenure security (RRI, 2020), they might still provide refuges for forest-dependent vertebrates. While this is encouraging, climate change could exacerbate the extinction risks of these species, in particular ectothermic amphibians and reptiles (herpetofauna; Mi et al., 2023). While Mi et al. examined the importance of PAs as refuges for herpetofauna under current and future climate scenarios, the additional range coverage of IPLs may provide vital buffer for at-risk species.

4.5.3. Limitations and conclusion

While the IPL dataset used represents the most comprehensive assessment of terrestrial lands where Indigenous Peoples have customary ownership, management and/or governance arrangements in place, several limitations suggest caution in interpretation. The dataset is based on a particular definition of Indigenous Peoples (see Garnett et al., 2018) and is certainly incomplete as the available maps varied in quality and were likely to have been deficient in countries where publicly available data are limited. However, differences in areas mapped between the IPL dataset of Garnett et al. (2018) and those estimated by RRI (2020) as belonging to Indigenous and local communities, which is greatest in Africa (27.7% versus 69.5% of geographical area, respectively), can be explained by the latter map conflating Indigenous Peoples with Local Communities, a practice now considered undesirable (IIPFCC, 2022).

We used Area of Habitat based on species distribution range maps rather than actual presence or absence of species or models based on those. This risks commission errors where species are considered present where they are not, particularly at the relatively high resolution of this study (Di Marco et al., 2017), though we minimised this by identifying forest-dependent species and using recent forest cover to obtain AOH. We also used the Global Forest Change dataset (Hansen et al. 2013) tree cover in 2020 at 50% threshold to determine where tropical forests remain in 2020. This dataset is known to underestimate deforestation and degradation (Vancutsem et al., 2021), as such our definition of remaining forests in 2020 may be overestimated which correspondingly increases the AOH of forest-dependent vertebrates. Our comparison of IPLs with their 10-km buffer zones also did not account for potential localised spillover effects adjacent to IPLs, since deforestation leakage from IPLs would reduce species' AOH outside of IPLs and may thus not present a complete understanding of IPLs and their importance for forest-dependent vertebrate diversity. There may also be gaps in the range maps, particularly in IPLs, since such areas are often surveyed less well than other lands (dos Santos et al., 2015). Research permits to sample biodiversity in many countries do not include authorisation to enter IPLs (e.g. Bolivia and Brazil), and legislation controlling access to IPLs may be a potential barrier for carrying out conservation-related research there (dos Santos et al., 2015). As such, while we present a comprehensive overview of vertebrate diversity in IPLs across the tropics, these data limitations should be taken into consideration.

Future research should thus focus on using survey data alongside ethnographic and participatory methods (Noss & Leny Cuellar, 2008) to better understand species abundance and distributions within IPLs, especially of larger-bodied species at risk of overhunting. These can help introduce sustainable hunting quotas or community-imposed bans on sensitive species. We focused on four major vertebrate taxa, which are good indicators of patterns in other taxa (Barlow et al., 2007; Edwards et al., 2014). However, there remain many gaps in taxonomic coverage. For example, it would be valuable to know whether IPLs are associated with insect diversity given Indigenous peoples' biocultural approaches to pollinator conservation (R. Hill et al., 2019), the crucial role that insects play in ecosystem functioning (Ewers et al., 2015) and their drastic global declines in diversity and abundance (Forister et al., 2019).

Nonetheless, our findings suggest that IPLs and their traditional stewards are critical for maintaining vertebrate biodiversity across the tropics. These results strongly align with those of the of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services Global Assessment (IPBES, 2019) and other global studies on Indigenous land-based stewardship (ICCA Consortium, 2021). Concerted action to support Indigenous Peoples in securing their lands and recognition of their historical rights to do so is thus inextricably linked to global efforts to combat biodiversity loss.

Our findings can support decision-making of where and how conservation interventions could occur, specifically the kinds of land management or ownership agreements that Indigenous Peoples can negotiate for to contribute to national conservation targets, should they wish to do so (Renwick et al., 2017). Considering that land tenure insecurity is pervasive across much of the tropics (Ceddia et al., 2015; Robinson et al., 2014), granting Indigenous Peoples formal legal title to their lands should be seen as an important mechanism for protecting IPLs from encroachment and safeguarding the biodiversity they harbour (Baragwanath & Bayi, 2020; Blackman et al., 2017). Any conservation efforts taking place in IPLs must include the participation of Indigenous communities throughout the entire conservation planning process. Recognising the agency and leadership of Indigenous Peoples as rights-, knowledge-, and stake-holders (beyond consent-giving), and the need for equitable distribution of benefits and compensation for costs are vital to underpin the success of the newly established Kunming-Montreal Global Biodiversity Framework (Reyes-García et al., 2021; Sandbrook et al., 2023).

4.6. Acknowledgements

We thank IT Services at The University of Sheffield for the provision of services for High Performance Computing. We acknowledge Indigenous Peoples and traditional custodians worldwide and recognise their collective wisdom. Funding was provided to D.P. Edwards from the Natural Environment Research Council (grant number NE/R017441/1).

4.7. Conflicts of Interests

All authors declare no conflicts of interest.

4.8. Data and Code Availability

Original data used for analysis are publicly available or can be requested from the respective authors. All code used in this analysis and the final results datasets used to produce figures are provided in the corresponding author’s GitHub repository: https://github.com/JocelyneSze/PhD-Ch4-biodiversity

4.9. Supplementary Information

STable 1. Percentage cover of Indigenous Peoples' Land in country, limited to tropical and subtropical forest biomes extent.

ISO3	Name	IPL (%)	ISO3	Name	IPL (%)
	AFRICA			**ASIA**	
BDI	Burundi	18.79	BGD	Bangladesh	57.46
BEN	Benin	82.71	CHN	China	30.23
CAF	Central African Republic	43.69	IDN	Indonesia	58.01
CIV	Cote d'Ivoire	18.78	IND	India	56.49
CMR	Cameroon	39.69	KHM	Cambodia	59.23
COD	Republic of Congo	22.7	LAO	Laos	86.59
COG	Democratic Republic of the Congo	19.68	LKA	Sri Lanka	1.43
ETH	Ethiopia	54.37	MMR	Myanmar	73.41
GAB	Gabon	46.55	MYS	Malaysia	70.96
KEN	Kenya	42.86	NPL	Nepal	72.77
NGA	Nigeria	27.14	PAK	Pakistan	66.18
RWA	Rwanda	25.34	PHL	Philippines	18.81
TGO	Togo	2.82	THA	Thailand	77.11
TZA	Tanzania	1.67	TWN	Taiwan	26.76
UGA	Uganda	3.27	VNM	Vietnam	75.93
REGIONAL AVERAGE		30	REGIONAL AVERAGE		55.42
	AMERICA			**OCEANIA**	
ARG	Argentina	26.34	AUS	Australia	12.22
BLZ	Belize	44.93	NCL	New Caledonia	44.23
BOL	Bolivia	35.04	REGIONAL AVERAGE		28.22
BRA	Brazil	31.03			
COL	Colombia	40.97			
CRI	Costa Rica	17.78			
DMA	Dominica	5.64			
ECU	Ecuador	40.42			
GTM	Guatemala	70.66			

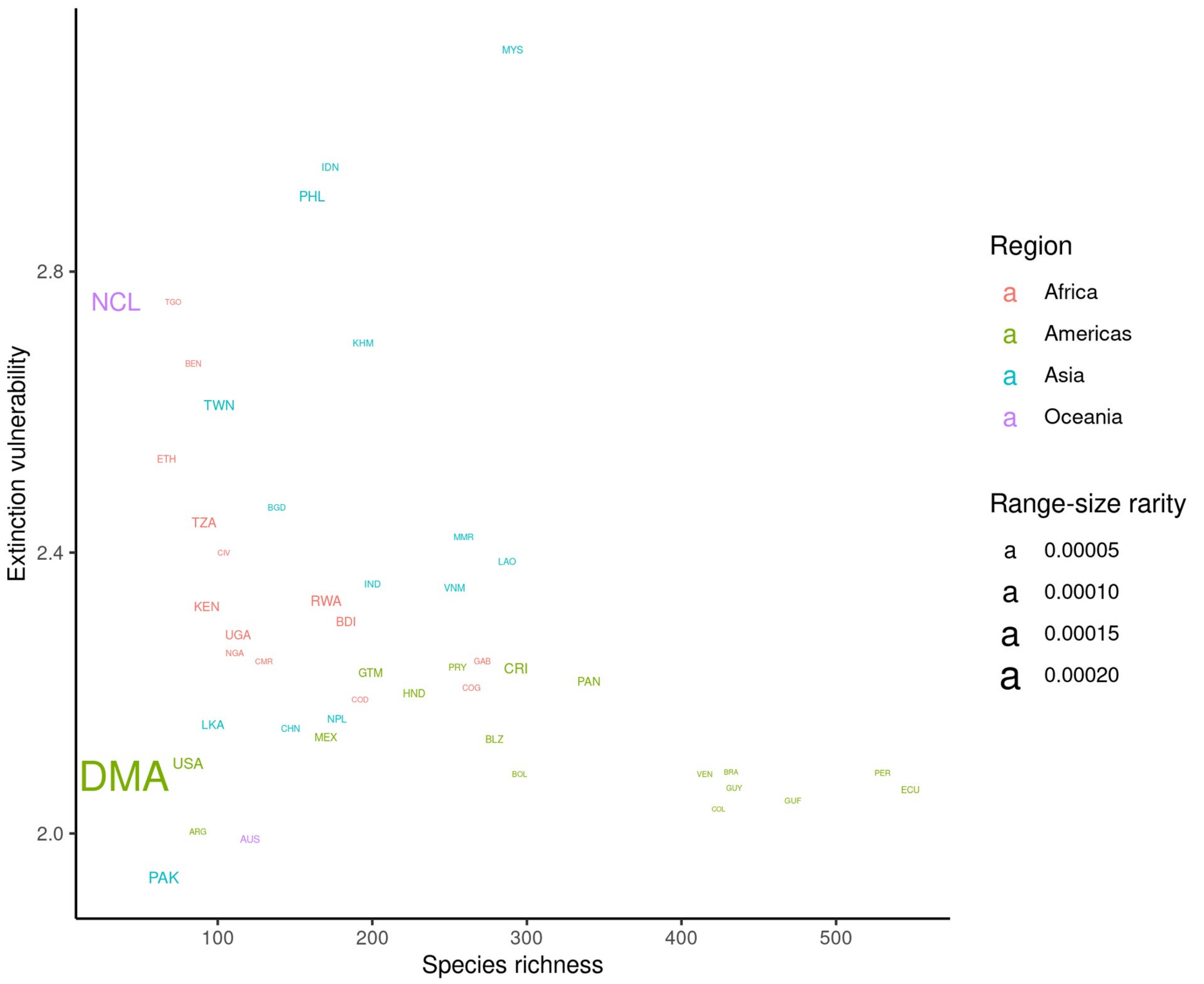
GUF	French Guiana	15.47
GUY	Guyana	15.96
HND	Honduras	39.72
MEX	Mexico	37.68
NIC	Nicaragua	52.62
PAN	Panama	55.2
PER	Peru	36.67
PRY	Paraguay	1.46
SLV	El Salvador	33.06
SUR	Suriname	42.04
USA	United States of America	0.65
VEN	Venezuela	72.76
REGIONAL AVERAGE		34.1

	Species Richness					Extinction Vulnerability					Range-size rarity				
Country	Min	Tukey LH	Median	Tukey UH	Max	Min	Tukey LH	Median	Tukey UH	Max	Min	Tukey LH	Median	Tukey UH	Max
AFRICA															
BDI	140	166	178	189	206	2.3	2.4	2.4	2.5	2.6	1.4E-06	3.8E-06	5.8E-06	9.9E-06	1.5E-05
BEN	62	75	79	82	91	2.4	2.7	2.8	2.8	2.9	5.4E-07	6.1E-07	6.1E-07	6.2E-07	1.7E-06
CAF	51	78	90	116	301	2.0	2.1	2.1	2.4	2.6	2.3E-07	3.1E-07	3.3E-07	3.7E-07	5.9E-06
CIV	55	82	99	125	142	2.1	2.4	2.5	2.6	2.8	3.2E-07	4.0E-07	4.6E-07	5.4E-07	6.2E-07
CMR	40	89	125	301	341	2.0	2.3	2.3	2.4	3.2	2.4E-07	3.8E-07	4.8E-07	8.5E-07	6.9E-04
COD	23	101	187	219	334	2.0	2.2	2.3	2.4	3.0	2.4E-07	3.5E-07	4.4E-07	5.6E-07	2.0E-03
COG	178	249	259	274	320	2.1	2.3	2.3	2.4	2.4	3.7E-07	5.1E-07	5.7E-07	6.3E-07	9.8E-07
ETH	23	55	62	68	84	2.4	2.6	2.6	2.7	3.6	4.4E-07	9.2E-07	1.9E-06	3.1E-06	4.3E-04
GAB	195	254	266	283	327	2.1	2.3	2.3	2.4	2.4	4.0E-07	5.4E-07	6.0E-07	7.0E-07	4.4E-06
KEN	35	64	88	112	180	2.1	2.2	2.4	2.8	3.1	6.6E-07	4.1E-06	6.4E-06	7.4E-06	1.2E-03
NGA	14	87	106	122	219	2.2	2.3	2.4	2.4	4.1	2.7E-07	4.5E-07	8.3E-07	1.6E-06	2.1E-05
RWA	62	145	165	189	222	2.1	2.4	2.4	2.5	3.1	2.1E-06	7.7E-06	9.3E-06	1.0E-05	5.7E-05
TGO	63	65	66	69	83	2.4	2.5	2.9	2.9	2.9	3.6E-07	3.7E-07	4.3E-07	6.2E-07	1.7E-06
TZA	46	77	86	98	129	2.4	2.5	2.5	2.6	3.0	4.2E-07	2.7E-06	7.8E-06	1.3E-05	2.6E-05
UGA	47	77	108	191	263	2.0	2.3	2.4	2.5	3.1	4.0E-07	2.5E-06	5.4E-06	9.1E-06	5.8E-05
AMERICAS															
ARG	20	71	82	186	287	2.0	2.1	2.1	2.3	3.4	4.4E-07	5.1E-07	6.5E-07	1.9E-06	4.1E-05

BLZ	230	238	274	276	284	2.2	2.2	2.2	2.2	2.3	1.5E-06	1.8E-06	1.8E-06	2.0E-06	4.7E-05
BOL	18	200	290	380	538	2.0	2.2	2.2	2.2	2.5	2.5E-07	3.4E-07	4.0E-07	5.2E-07	5.2E-05
BRA	15	404	427	485	572	2.1	2.2	2.2	2.3	3.4	2.4E-07	2.9E-07	3.2E-07	3.5E-07	3.2E-04
COL	34	335	419	443	541	2.0	2.1	2.1	2.2	4.0	2.0E-07	2.8E-07	3.1E-07	3.5E-07	1.0E-03
CRI	9	224	288	321	383	2.2	2.3	2.3	2.4	3.7	2.4E-06	6.5E-06	1.0E-05	1.6E-05	1.6E-03
DMA	32	33	34	35	38	2.2	2.2	2.2	2.2	2.6	1.2E-04	2.1E-04	2.2E-04	2.3E-04	3.7E-04
ECU	20	442	543	555	592	2.0	2.2	2.2	2.2	3.4	4.3E-07	8.3E-07	1.2E-06	2.5E-06	8.6E-05
GTM	42	164	194	267	289	2.0	2.2	2.3	2.5	3.1	1.5E-06	2.2E-06	4.0E-06	1.1E-05	5.1E-03
GUF	120	461	467	472	480	2.1	2.1	2.1	2.1	2.2	3.6E-07	4.3E-07	5.1E-07	5.3E-07	3.0E-06
GUY	130	401	429	449	484	2.1	2.2	2.2	2.2	2.4	3.0E-07	3.6E-07	3.9E-07	5.6E-07	3.5E-04
HND	1	188	222	286	303	2.0	2.2	2.3	2.3	4.6	3.6E-07	2.0E-06	2.7E-06	3.9E-06	1.0E-02
MEX	1	125	165	202	276	2.0	2.2	2.2	2.3	3.7	6.1E-07	2.1E-06	3.1E-06	5.8E-06	3.2E-03
NIC	41	159	294	305	340	2.2	2.3	2.3	2.4	2.6	9.0E-07	1.9E-06	2.3E-06	2.6E-06	8.1E-05
PAN	1	273	335	344	437	2.0	2.2	2.3	2.4	4.9	1.7E-06	3.3E-06	4.7E-06	8.8E-06	6.4E-04
PER	4	402	525	546	593	2.0	2.1	2.2	2.2	4.3	3.0E-07	4.5E-07	5.5E-07	9.7E-07	6.4E-03
PRY	74	229	250	259	274	2.1	2.3	2.3	2.4	2.4	4.8E-07	1.2E-06	1.2E-06	1.3E-06	3.2E-06
SLV	89	149	157	161	198	2.2	2.2	2.2	2.3	2.6	2.1E-06	3.1E-06	3.3E-06	3.6E-06	1.1E-04
SUR	114	452	460	467	480	2.1	2.1	2.1	2.2	2.2	3.4E-07	4.0E-07	4.3E-07	4.5E-07	2.9E-06
USA	1	68	76	79	87	2.0	2.2	2.2	2.2	10.6	9.2E-06	1.2E-05	1.3E-05	1.5E-05	2.1E-03
VEN	2	340	410	437	478	2.0	2.2	2.2	2.2	4.0	2.6E-07	3.6E-07	4.3E-07	7.2E-07	3.4E-04
								ASIA							
BGD	104	122	133	141	184	2.4	2.5	2.6	2.6	2.8	5.3E-07	6.3E-07	6.6E-07	6.9E-07	1.0E-06

CHN	7	107	142	171	300	2.0	2.2	2.3	2.4	2.8	4.0E-07	7.2E-07	1.0E-06	2.6E-06	1.0E-03
IDN	0	134	168	254	313	0.0	2.4	3.0	3.4	6.7	0.0E+00	1.4E-06	2.2E-06	5.6E-06	7.3E-03
IND	1	117	195	235	301	2.0	2.3	2.5	2.5	9.0	7.9E-08	9.5E-07	1.7E-06	2.1E-06	1.5E-03
KHM	127	177	189	204	259	2.4	2.6	2.8	3.0	3.4	7.4E-07	1.2E-06	1.4E-06	2.0E-06	4.1E-05
LAO	133	259	282	292	323	2.1	2.4	2.5	2.6	3.3	6.4E-07	9.3E-07	1.2E-06	1.5E-06	6.4E-04
LKA	86	91	92	93	102	2.2	2.2	2.3	2.3	2.3	3.9E-06	4.9E-06	5.2E-06	5.3E-06	6.8E-06
MMR	24	218	254	269	309	2.1	2.4	2.5	2.6	3.1	4.9E-07	7.5E-07	8.9E-07	1.2E-06	8.2E-05
MYS	5	266	286	299	346	2.2	3.1	3.2	3.3	3.8	9.7E-07	1.8E-06	2.2E-06	3.1E-06	4.7E-03
NPL	44	153	172	193	253	2.0	2.2	2.3	2.3	2.7	9.2E-07	1.4E-06	2.5E-06	3.8E-06	1.4E-05
PAK	22	51	60	75	98	2.0	2.0	2.0	2.1	2.4	3.7E-06	1.1E-05	1.6E-05	1.8E-05	3.7E-04
PHL	11	142	156	164	187	2.1	2.8	3.0	3.2	3.6	1.2E-06	7.0E-06	9.0E-06	1.4E-05	1.8E-03
THA	3	227	258	280	314	2.3	2.5	2.5	2.7	8.7	5.5E-07	9.1E-07	1.1E-06	1.4E-06	1.7E-03
TWN	54	88	96	104	113	2.3	2.5	2.7	2.8	3.0	3.3E-06	6.4E-06	8.4E-06	1.3E-05	1.8E-04
VNM	136	222	248	277	314	2.1	2.4	2.5	2.7	3.3	5.7E-07	1.4E-06	1.8E-06	3.2E-06	2.0E-03
								OCEANIA							
AUS	5	85	116	155	174	2.0	2.1	2.1	2.1	2.5	7.5E-07	1.6E-06	2.3E-06	1.1E-05	1.7E-03
NCL	0	17	29	32	38	0.0	2.7	2.9	3.7	8.0	0.0E+00	4.7E-05	6.0E-05	8.9E-05	5.4E-03

STable 2. Minimum, Tukey lower hinge, median, Tukey upper hinge, and maximum values of species richness, extinction vulnerability, and range-size rarity for each country.



SFigure 1. Median value of species richness, extinction vulnerability, and range-size rarity of IPLs in the 53 countries with tropical forests in 2020.

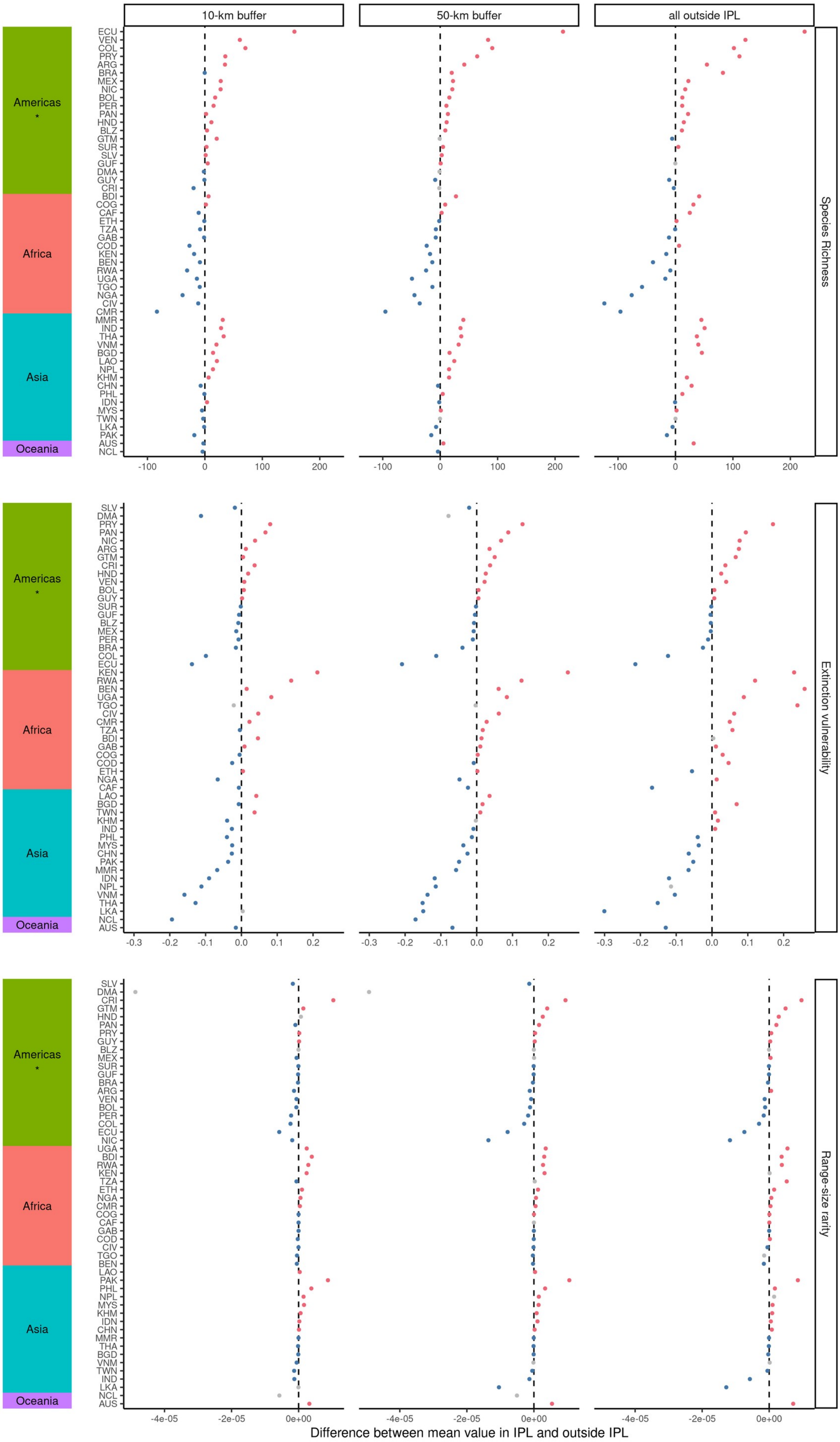
Country-level permutation tests

We conducted country-level permutation tests to test the significance in difference between conservation metrics inside and outside IPLs. We constructed null distributions by first cropping conservation metric rasters (species richness, extinction vulnerability, and range-size rarity, for all taxa and each taxon separately) to the country vector file and extracting the values within IPLs, the 10-km buffer, 50-km buffer, and rest of the country in a data frame. We filtered the country data frame for values within IPLs and the 10-km buffer, then randomly drew without replacement for values equivalent to the number of IPL pixels for that country, and calculated the differences in mean of selected and non-selected values. This procedure of randomly drawing without replacement was repeated 1000 times to create the null distribution. We then ran two-tailed tests between the observed difference and the null distribution of differences.

To obtain the null distribution of differences between inside IPLs and the 10-km buffer zone,

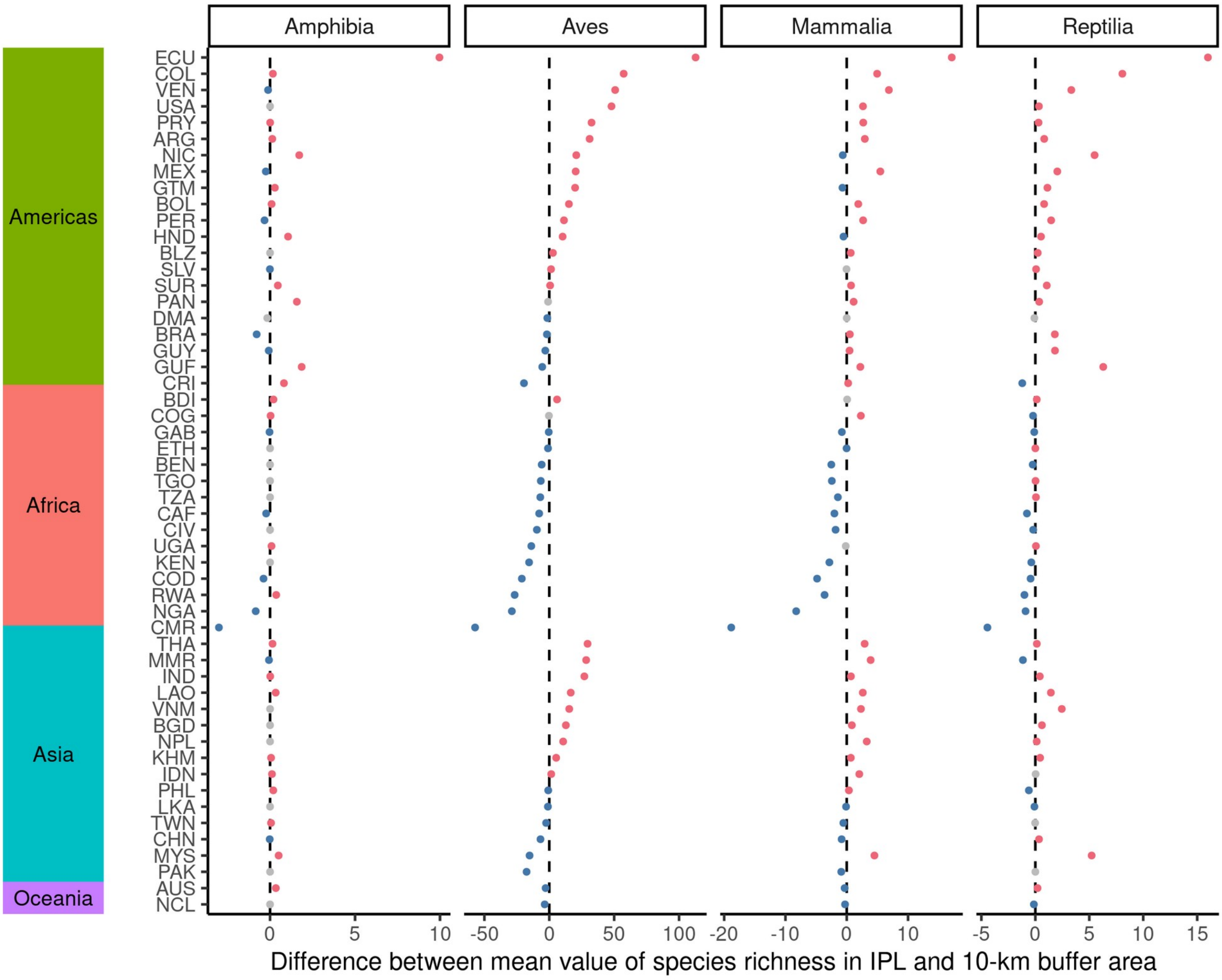
For all taxa (i.e. the conservation metric raster for all vertebrates combined), we conducted this permutation test for IPLs and the 10-km buffer, 50-km buffer, and all outside IPL, while for each separate taxon, we conducted this permutation test only for IPLs and the 10-km buffer.

For relevant R code, please see: https://github.com/JocelyneSze/PhD-Ch4-biodiversity under the Code folder, files 11-Country_bootstrapping.R and 12-Country_analysis.R

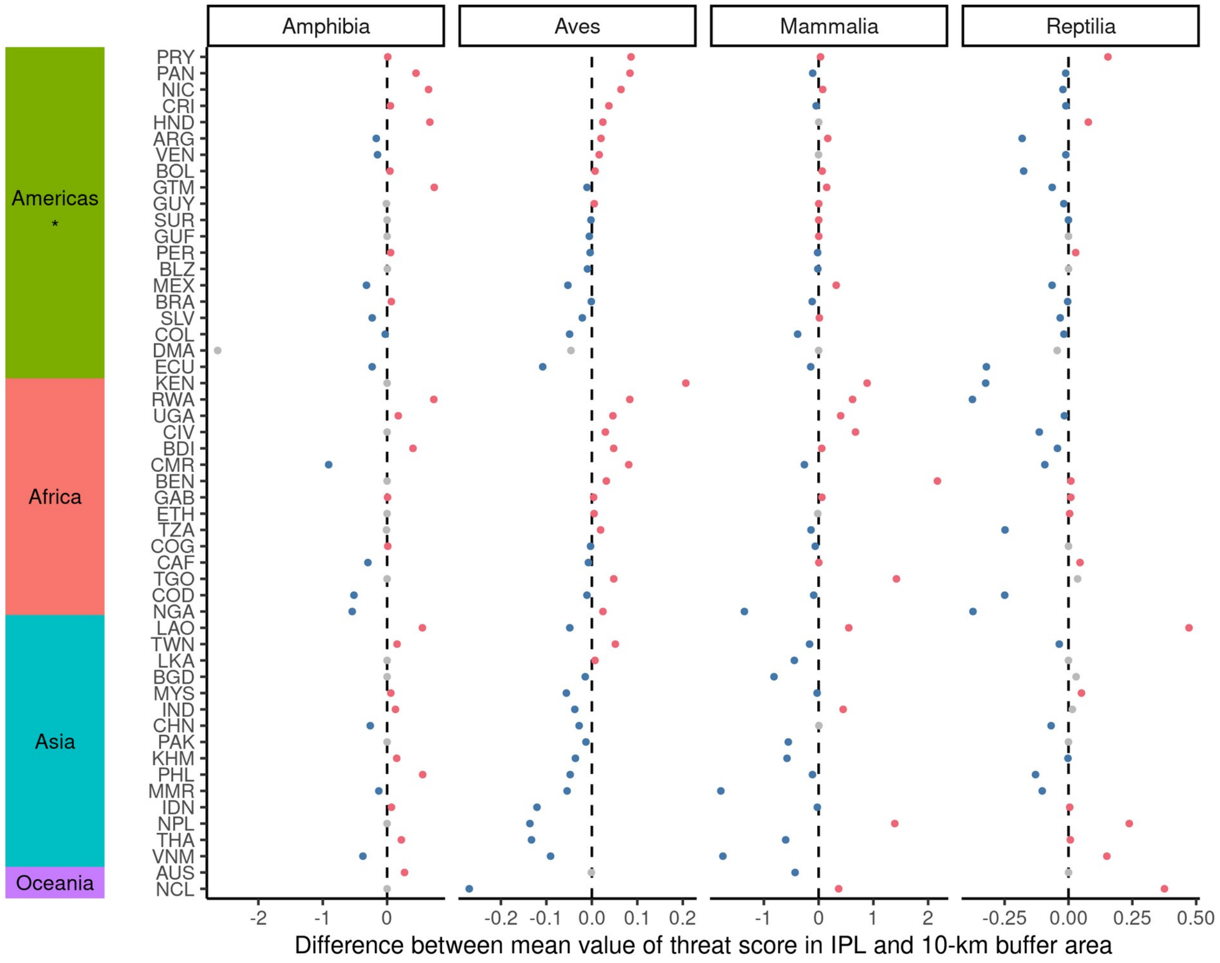

10-km buffer
50-km buffer
all outside IPL
Species Richness
Extinction vulnerability
Range-size rarity
Americas
Africa
Asia
Oceania
Difference between mean value in IPL and outside IPL

*SFigure 2. Mean difference between values of species richness, extinction vulnerability, and range-size rarity inside IPLs and 10 km buffer, 50 km buffer, and all areas outside of IPLs. Red dots represent significantly positive difference (greater value inside IPL), blue dots represent significantly negative difference (smaller value inside IPL), and grey dots represent no significant difference, at alpha=0.05 level. *USA is not plotted here for visual purposes as an outlier; Values for USA are as follows:*

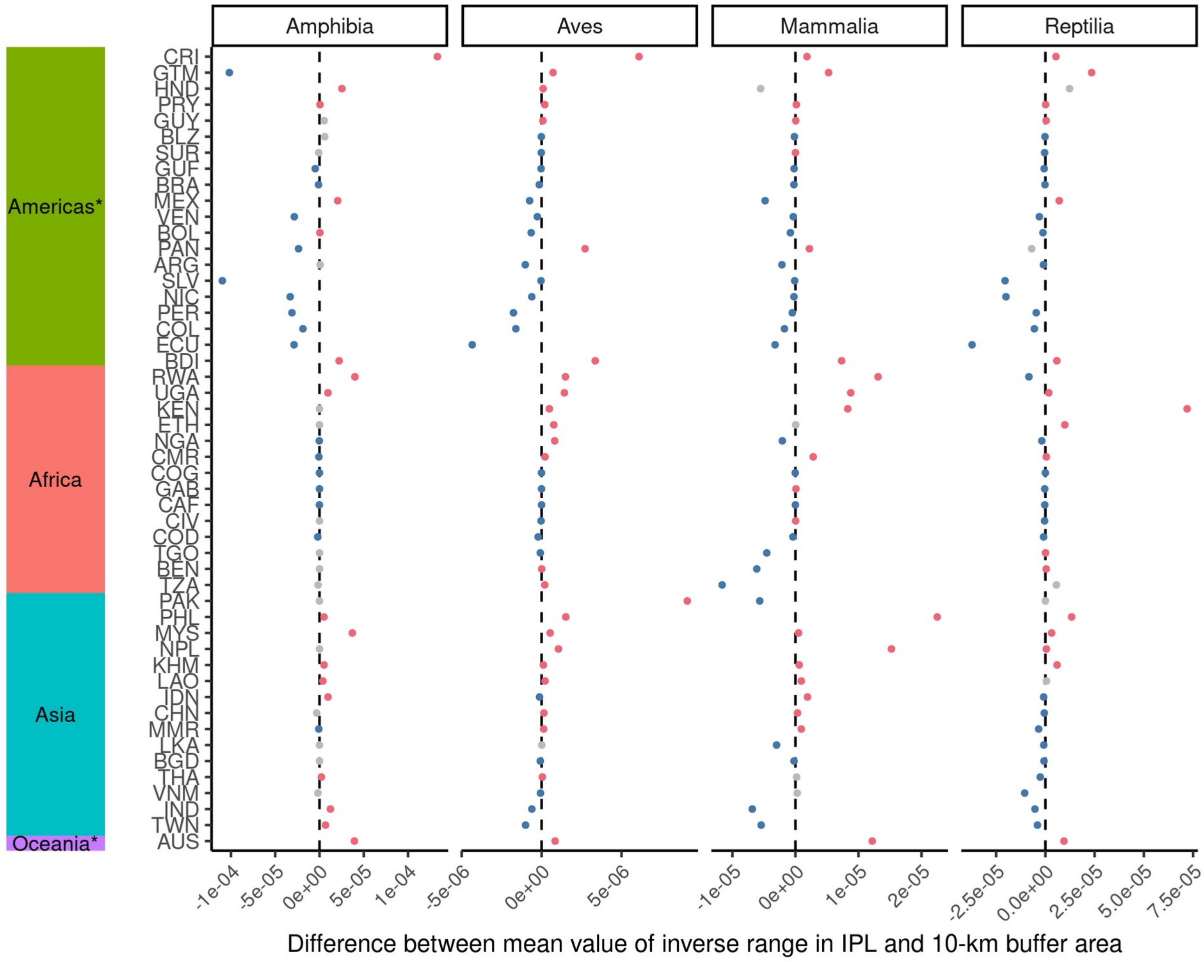
Difference between IPLs and outside areas for USA	**10 km buffer**	**50 km buffer**	**All areas outside**
Species richness	50.9 (sig.pos.)	40.0 (sig.pos.)	14.2 (sig.pos.)
Extinction vulnerability	-1.88 (sig.neg.)	-1.55 (sig.neg.)	0.301 (sig.pos.)
Range-size rarity	-0.0003 (sig.neg.)	-0.0003 (sig.neg.)	0.00004 (sig.pos.)



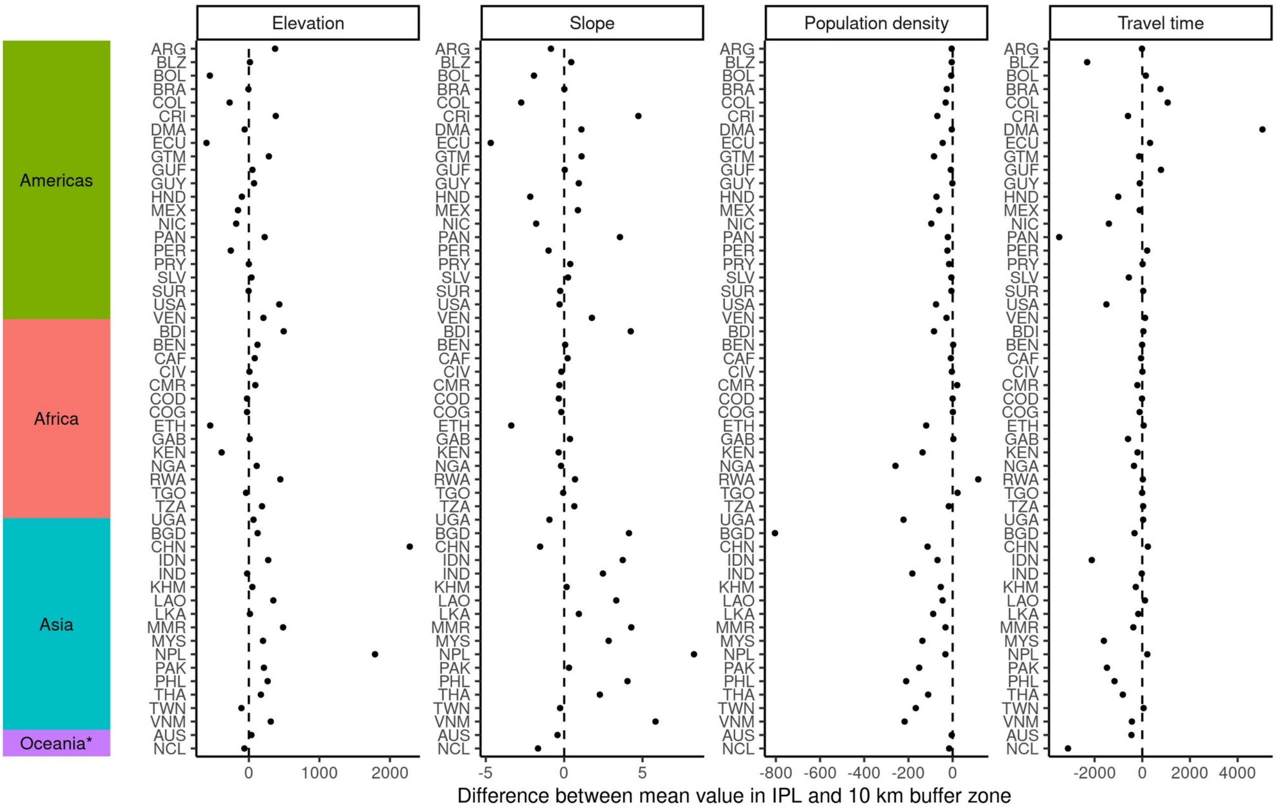
*SFigure 3. Difference between mean values of **species richness** within IPL and the 10 km buffer area outside for amphibians, birds, mammals, and reptiles. Red dots represent significantly positive difference (greater value inside IPL), blue dots represent significantly negative difference (smaller value inside IPL), and grey dots represent no significant difference, at alpha=0.05 level.*



*SFigure 4. Difference between mean values of **extinction vulnerability** within IPL and the 10 km buffer area outside for amphibians, birds, mammals, and reptiles. Red dots represent significantly positive difference (greater value inside IPL), blue dots represent significantly negative difference (smaller value inside IPL), and grey dots represent no significant difference, at alpha=0.05 level. *USA is not plotted here for visual purposes as an outlier; values for amphibians=0(not sig.), birds=-1.87(sig.nev.), mammals=1.43(sig.pos.), reptiles=0.66(sig.pos.)*



SFigure 5. Difference between mean values of ***range-size rarity*** *within IPL and the 10 km buffer area outside for amphibians, birds, mammals, and reptiles. Red dots represent significantly positive difference (greater value inside IPL), blue dots represent significantly negative difference (smaller value inside IPL), and grey dots represent no significant difference, at alpha=0.05 level. *USA, DMA, and NCL are not plotted here for visual purposes as outliers; values for USA: amphibians=0(not sig.), birds=-0.0003(sig.neg.), mammals=0.00002(sig.pos.), reptiles=0.0000007(sig.pos.); for DMA: amphibians=-0.0009(not.sig.), birds=-0.00003(not.sig.), mammals=0(not.sig.), reptiles=0.0003(sig.pos.); for NCL: amphibians=0(not.sig.), birds=0.000009(sig.pos.), mammals=-0.000002(sig.neg.), reptiles=-0.00036(sig.neg.)*



SFigure 6. Difference between mean values within IPL and the 10 km buffer area outside for elevation, slope, population density, and travel time to nearest city with ≥5000 population.

Chapter 5

General Discussion

5.1. Summary

International area-based conservation policies are increasingly acknowledging and including the contributions of Indigenous peoples and local communities (CBD, 2022). Whilst being a positive development for ensuring that Indigenous peoples' rights are not infringed by conservation, there is still a need to establish that these alternative area-based conservation measures are effective in delivering conservation outcomes. However, though there have been studies at local and regional scales, there have not been global studies evaluating the conservation value of Indigenous peoples' lands. This thesis thus set out to fill this gap using big data and robust statistical methods at the pan-tropical scale.

In Chapter 2, I used the European Commission's Joint Research Centre (ECJRC) Tropical Moist Forest dataset (Vancutsem et al., 2021) and the Global Forest Watch (GFW) Tree Cover Loss dataset (Hansen et al., 2013) to evaluate deforestation and degradation rates from 2010-2019 in Indigenous Lands (ILs), protected areas (PAs), the spatial overlap of protected areas in Indigenous lands (PIAs), and non-protected areas. Results from the ECJRC data showed that deforestation (permanent conversion from tree cover to other) and degradation (visible disturbance in tree cover canopy lasting <2.5 years) rates were lower in ILs compared to matched counterfactual non-protected areas across the tropics. The GFW data also showed reduced deforestation rates in ILs compared to matched counterfactual non-protected areas, with greater avoided deforestation in ILs in Africa and Americas but lesser avoided deforestation in ILs in Asia, compared to avoided deforestation using ECJRC data.

The GFW dataset is widely used, having been publicly available since 2013, while the ECJRC dataset had been available only since 2021. Although both use the same Landsat data as the source, the processing methods are different (Vancutsem et al., 2021) and my findings suggest that short-term degradation might appear as deforestation in the GFW data. Focusing on ECJRC data, I found that in Africa, ILs have lower deforestation and degradation rates compared to PAs, while in the Americas, PAs consistently have lower deforestation and degradation rates than ILs, and in Asia, deforestation rates are lower in PAs but degradation rates are lower in ILs. In PIAs in the Americas and Asia, deforestation and degradation was reduced relative to non-protected areas; in the Americas, they perform similarly to ILs, while in Asia, they reduced more deforestation and degradation than both ILs and PAs. However, PIAs in Africa appear to have little or worse effect on deforestation and degradation rates compared to non-protected areas.

As deforestation and degradation rates show only recent direct impacts, I used the Forest Landscape Integrity Index (Grantham et al., 2020) in Chapter 3 to provide a more comprehensive perspective on the state of tropical forests. Additionally, given the long-term but low-intensity human use and influence in tropical forests (Roberts et al., 2017), I used the Anthromes dataset (Ellis et al., 2021) to understand how land-use intensity in forests had changed between 1950 and 2010. I found that ILs in Americas had the same forest integrity score as non-protected areas, and ILs in Asia had worse forest integrity scores than non-protected areas, and only in Africa did ILs have better forest integrity scores than non-protected areas. Compared to PAs, ILs all had lower forest integrity scores. However in PIAs, forest integrity scores were the highest. Land-use intensity was also lowest in PIAs both in 1950 and 2010, although there has been an increase in land-use intensity across all categories.

It was interesting that PIAs appeared to provide the best conservation outcome in Chapter 3, while ILs generally had poor outcomes and consistently worse than PAs. Yet in Chapter 2, ILs had positive conservation outcomes and in Americas and Africa, better than PAs, while PIAs were variable. Part of this mismatch may be due to the dependent variables used to measure conservation outcomes; the Forest Landscape Integrity Index includes observed pressures on forests such as deforestation, but also inferred pressures modelled based on observed pressures, and lost forest connectivity (Grantham et al., 2020). Taking the results of Chapters 2 and 3 together might suggest that there has been anthropogenic modification in ILs over the past decades, while PAs have generally managed to avoid most of this anthropogenic modification, which corroborates earlier studies using other measures of anthropogenic pressures (Geldmann et al., 2019). However, in the last decade, ILs have managed to reduce the amount of direct forest loss occurring. ILs that are overlapped by PAs have also avoided anthropogenic modification across the tropics, yet failed to reduce forest loss and degradation in Africa.

In Chapter 4, I focused on the importance of ILs for forest-dependent vertebrates. Using the distributional range files of 11,872 species of forest-dependent amphibians, birds, mammals, and reptiles, I modified the range files to the remaining tropical forest extent in 2020 and the elevational ranges of the species to approximate their Areas of Habitat. More than three-quarters of tropical forest-dependent vertebrates had part of their ranges within ILs. I evaluated the additionality contributed by ILs to the ranges of these species, finding that 288 species had 100% of their remaining habitat within ILs, including areas overlapped by PAs, and 120 species were found only in ILs, exclusive of PAs. Using three metrics of species richness, extinction vulnerability, and range-size rarity, I calculated the values within ILs for 53 countries and the 10 km buffer outside of ILs. Following general global species richness trends, species richness in ILs were concentrated in South Americas, contributed largely by birds. Extinction vulnerability of vertebrates in ILs was focused in Southeast Asia, while range-size rarity was most important for the small island countries of Dominica and New Caledonia. Compared to outside ILs however, species richness was greater across most countries in Americas and Asia but not in Africa. Extinction vulnerability scores were lower in ILs compared to outside across most countries, and there was little difference in range-size rarity scores inside and outside ILs. These findings highlight the extent to which Indigenous lands provide habitat for biodiversity, particularly potentially as refuges from wider habitat loss outside.

Overall, my three Chapters suggest that ILs have an important role to play in contributing to conservation outcomes, be it reducing deforestation, maintaining forest integrity, or providing habitat for tropical forest-dependent vertebrates.

5.2. Methodological limitations of measuring contributions

The methods used in this thesis, in particular, the statistical matching and regression in Chapters 2 and 3, are generally considered state-of-the-art and provide robust estimations for deforestation, degradation, and forest integrity in ILs, PAs, PIAs, and non-protected areas. Nonetheless, there are still limitations to the methodology and philosophical implications to be considered behind the framing of research questions.

First, in my chapters, I chose to use particular time frames: deforestation and degradation from 2010 to 2018/2019 in Chapter 2, forest integrity in 2019 and Anthrome states in 1950 and 2010 in Chapter 3, and forest extents in 2020 in Chapter 4. The choice of cut-off year was partly due to data availability and partly due to a bias for round numbers. As such, my results reflect a snapshot in time and results are likely to vary across time. For example, certain political events such as the election of a strongly neoliberal government with little environmental regard are known to influence deforestation (Burgess et al., 2012; Ruggiero et al., 2021). Choosing different time frames might thus result in different estimates, though this difference might be slight unless a significant number of countries within a given region hold elections in the same year. Additionally, the outcomes (deforestation/degradation) are likely to change over time and these changes may be at different rates within ILs, PAs, PIAs, and non-protected areas. Conducting a Difference-in-Difference study design that additionally estimates effects over time would mitigate some of these concerns, however, there lies the trade-off with computational tractability/efficiency.

Second, the epistemological assumptions behind quasi-experimental methods hold that any difference between the treatment (ILs, PAs, or PIAs) and the matched counterfactual (non-protected areas) is the measurement of value. Particularly when conducted across large geographical scales as I had done, this risks overlooking nuances in spatial contexts and interpretation of results. For example, in Chapter 2, ILs in French Guiana had 90% lower deforestation compared to matched non-protected areas, however, the non-protected area deforestation rates were low to begin with (0.002±1.6E-5). In contrast, in Belize, deforestation rates in ILs were 31% lower than matched non-protected areas but deforestation in both ILs and non-protected areas were at least 3 times higher than regional average of 0.003±0.008. Small values of deforestation rates are more sensitive to percent changes which may overly-amplify contrasts. Defining what should be considered a 'successful' outcome requires careful thought for the specific context, which are difficult to achieve using large datasets and statistical methods that emphasise standardisation. As such, while statistical matching and counterfactual thinking have their value, it is crucial that they are evaluated in appropriate contexts with a view of the overall picture and the goals that conservation aspires to achieve. In particular, such methodology may be more suitable and appropriate for measuring overall losses (e.g. tree cover loss) or outcomes that are more amenable to homogenisation, as opposed to specific species' loss and where targets are non-commensurable. It is essential for conservation to consider what we are wanting to conserve, what measurements are needed, whether these are specific to space and/or time, and how to achieve it (Redford et al., 2003).

Given these methodological considerations of using statistical methods to identify spaces that are more comparable considering location biases (yet not fully accounting for other political, economic, social, and ecological situations) in order to identify the adjusted-for contributions of ILs, PAs, and PIAs, I adopted a different philosophical approach in Chapter 4. Particularly since the outcome being measured was species diversity (as opposed to forests, which, at least with regards to measuring environmental outcomes, have become accepted as commensurable), I felt that applying the same approach of spatial matching would not suffice. First, because it would not be possible to fully account for factors influencing species diversity distributions (which are not the same factors that influence forest loss), and second, that it would be more informative to understand where species remain presently, rather than evaluate for counterfactual outcomes of what might be.

5.3. Implications for biodiversity conservation and climate change

Area-based conservation measures remain one of the primary policy tools to address these crises, having evolved from being focused on exclusionary 'fortress conservation' to more inclusive and equitable approaches that acknowledge and include Indigenous communities that often inhabit or depend on important conservation areas (Chapter 1, section 1.1.2). This thesis set out to quantify the contributions of Indigenous peoples' lands to tropical forest conservation, finding that overall, Indigenous lands do reduce deforestation by 17-26% and degradation by 9-18% (Chapter 2); cover the ranges of 8874 tropical forest-dependent vertebrates, including 288 species whose remaining range fall entirely within Indigenous peoples' lands (Chapter 4); and where they overlap with PAs, high forest integrity is maintained (Chapter 3).

These findings add to the growing evidence of Indigenous communities' contributions to conservation (FPP et al., 2020; WWF et al., 2021). While I am unable to establish legal recognition or tenure security as the mechanism for Indigenous peoples' lands delivering conservation outcomes, since the Indigenous peoples' land dataset do not provide these information, other studies suggest that having secure land tenure improves forest outcomes by giving communities greater control (Chapter 2, section 2.4). Formal and informal, State and non-State support and recognition also facilitate the success of Indigenous-managed conservation areas, even furthering their rights and political influence outside (Tran et al., 2020).

Critically, compared to externally controlled conservation initiatives (15.7% of n=102 studies), 55.9% of internally controlled initiatives (n=59) reported positive social and ecological outcomes, with more Indigenous and locally controlled intiatives reported in Latin America (53.8% of 39 studies), than in Africa (28.3% of 60) and Asia (23.2% of 56). These underscore the importance of Indigenous communities' participation in management and governance and internal legitimacy of conservation initiatives (Dawson et al., 2021). Good inter-institutional collaborations and relationships based on trust, often developed over time through conflict resolution processes, intercultural understanding, transparent and timely communication, and respect for local rights, as well as recognition and integration of local ecological knowledges into law and policy at higher levels of governance were also necessary enabling external factors that contributed to positive social and ecological outcomes (Dawson et al., 2021).

Through sustainable self-regulation of resource use, concerted habitat restoration, assertion of collective territory to prevent external encroachment and resist commercial and extractive pressures, and the ability to maintain or adapt natural resource management institutions and stewardship practices in the face of economic, political, and environmental change, Indigenous and local communities were able to effectively conserve biodiversity (Dawson et al., 2021). Many Indigenous communities had also established conservation initiatives as a way to reclaim, restore, and/or revitalise Indigenous management practices and access in response to losses from colonial practices and discriminatory actions against Indigenous peoples (Tran et al., 2020).

While Indigenous peoples are able to contribute to stemming biodiversity decline and mitigating climate change by reducing tropical forest loss and protecting important habitat for biodiversity, these successes were still limited by the lack of funding, capacity, and State legislation to recognise Indigenous-managed conservation areas (Tran et al., 2020). In cases where locally controlled conservation failed to deliver positive social and ecological outcomes, national legislative frameworks and policies and interactions with State and external actors presented challenges by limiting Indigenous decision-making, dismantling local institutions, disempowering local communities, and actively supporting the unsustainable exploitation of resources within their territory, especially where power imbalances remained reflected in wider socio-political dimensions (Dawson et al., 2021; Tran et al., 2020).

Although on the whole, Indigenous lands, particularly where Indigenous peoples have legal rights and secure tenure over their territory, are generally associated with positive environmental and social outcomes, there are still instances where this may not be so. For example, in Costa Rica, I found that deforestation rates were nearly 2 times higher in ILs than non-protected areas (Chapter 2), with an average of 19.7 fewer forest-dependent vertebrate species in ILs than in the 10-km buffer zone outside (Chapter 4). My analysis was conducted at the national level, thus making it difficult to make more specific interpretations regarding the factors or mechanisms that lead to these outcomes. Nonetheless, considerable research had been conducted in the BriBri and Cabécar Indigenous reserves in Talamanca, where dung beetle and mammal species richness were greatest in forests, followed by indigenous traditional cocoa and banana agroforestry systems, and least in plantain monocultures (Harvey et al., 2006). These Indigenous reserves were delineated by the Costa Rican State in 1977, partly in response to the increasing dominance of banana plantations and railroad tracks that facilitated capitalist agricultural expansion in the 1900s (Ramirez Cover, 2017) Given the dominance of neoliberal development in the country, market-oriented conservation interventions such as eco-tourism, Payments for Ecosystem Services and export-oriented agroforestry systems were rolled out to tackle the environmental degradation arising from increasing intensive land-uses. Despite legal technicalities that these Indigenous reserves are to be managed by the BriBri and Cabécar peoples, in practice, conservation organisations and agencies still have retain influence over these spaces, encouraging the production of for-export and for-profit agricultural products (Ramirez Cover, 2017). Without a more grounded understanding of the Costa

Rican context and the specific dynamics that occur within ILs there, however, it remains difficult to interpret the reasons for why ILs may have worse environmental outcomes than non-protected areas.

Looking again at the big picture, given that Indigenous peoples' lands cover 38% of the Earth's terrestrial surface, there remains huge potential for bending the curve of biodiversity decline (Leclère et al., 2020) with the support and contributions of Indigenous peoples. The reductions in direct forest loss and degradation in tropical forests and maintenance of forest integrity where PAs overlap with Indigenous lands imply reductions in carbon emissions (Walker et al., 2020) and protection of habitat for tropical forest-dependent species. However, I am unable to ascertain the actual state of biodiversity in Indigenous lands, such as through alpha or beta diversity or how rates of species turnover or species abundances are changing, and how they compare with non-protected and protected areas, which is important in evaluating whether these alternative area-based conservation measures help stem biodiversity decline.

With the rhetoric shifting from Indigenous peoples being vulnerable and passive victims of environmental change to being resilient and adaptive to environmental challenges given appropriate support and enabling policies (Ford et al., 2020), this thesis demonstrates the contributions that Indigenous peoples have made and highlights their potential in taking a central role in decision-making in tropical forest conservation.

5.4. Indigenous-led conservation

From being displaced to make way for conservation, to being seen as stakeholders of conservation projects, to the current push to being acknowledged as rightsholders, Indigenous peoples have made strong progress in regaining their rights and autonomy (Reyes-García et al., 2021). Most international conservation organisations have now shifted in language to portray Indigenous peoples as 'conservation allies', and policy processes and protocols require adherence to UNDRIP principles (CIHR, 2014). However, the far-reaching colonial legacies of conservation persist to the present, particularly in societal and organisation structures that continue to uphold dominant and colonial ideologies, precluding truly transformative and equitable approaches (Kashwan et al., 2021).

The lack of attention to the difference between capitalist-motivated industrial extractive activities from human activities in general as drivers of biodiversity decline (Moranta et al., 2021), and the lack of recognition that nature has been enhanced by human activity (Levis et al., 2017; Roberts et al., 2017) also persist in perpetuating colonial conservation ideas. There is a risk therefore that conservation organisations are merely utilising the narrative of rights-based conservation to further their own ends while presuming that the 'benefits ... will eventually trickle down to people' (Witter & Satterfield, 2019). Similarly, State governments whose primary objective is to achieve their national area-based targets in fulfilment of CBD commitments under the more equitable option of OECMs may fail to fully respect the processes and meaningfulness behind recognition of Indigenous peoples' contributions to conservation and rights to their territory and autonomy (Zurba et al., 2019).

First, dominant perceptions of Indigenous peoples characterise and essentialise Indigenous peoples as ecologically noble savages', which diminishes their autonomy and rights to self-determination, and fails to fully appreciate the differences in their worldviews and understandings of reality and knowledge from mainstream views. Such simplification and characterisation may also promote the narrative that Indigenous rights are contingent upon fulfilling conservation outcomes (Witter & Satterfield, 2019). Further, while conservation organisations claim to work with Indigenous peoples, the frequent characterisation of spaces in which Indigenous peoples live in as 'wilderness' has been argued to de-humanise Indigenous peoples (Fletcher et al., 2021), especially if power imbalances that still exist between international conservation organisations and Indigenous peoples are not addressed.

As such, before arriving at equitable conservation, there is the need to understand power relations between conservation and Indigenous communities, and acknowledge and redress past injustices committed against Indigenous peoples by conservation or in the name of conservation (Shackleton et al., 2023). Such acknowledgements are necessary for trust-building (Saif et al., 2022) before institutional infrastructure to co-produce transformative governance approaches can be built, which requires foregrounding equity and rights-based support of Indigenous communities' agency, access, and decision-making autonomy (Armitage et al., 2020).

Lastly, as conservationists, we need to expand our understanding of what it means to do conservation, to respect and prioritise Indigenous knowledges and take on board their ways of understanding human-nature relationships (Fletcher et al., 2021; Obura et al., 2021; Witter & Satterfield, 2019). Typically, conservation might make use of Indigenous knowledge superficially and fail to deeply engage with Indigenous perspectives and worldviews. For example, in Sarawak, Malaysia, conservationists used Indigenous Iban's stories of Bornean orangutans (*Pongo pygmaeus*) in their conservation education programmes with Iban youths to reinforce and inculcate colonial ideas of species protection, while Iban's different classifications of maias (the Iban name for orangutans) and understanding of Iban and non-human species kinships, mutual responsibility, and reciprocity were neglected (Rubis, 2020).

Indigenous movements, particularly in Latin America, have focused not just on gaining legal recognition, representation, and land rights, but additionally advocated for their different conceptions of what constitutes a 'good life', which encompass principles of reciprocity, collective participation, social justice, and harmony with Mother Nature and with the community and family (ECLAC, 2014). Such conceptions are often in contrast to mainstream society's which is based on capitalist extractive development that see nature as objects to be exploited and reinforces the separation of human and nature (Escobar, 2011). Indigenous-led conservation may thus look very different from mainstream conservation, but through equitable collaborations, more effective conservation impacts may result (Ban et al., 2018).

While these recommendations are broadly aimed at conservation (organisations, researchers, practioners, and policy-makers) to challenge the existing power dynamics and set a more welcoming stage for Indigenous-led conservation, there are other practical considerations. Even with more supportive and enabling structures, initiative from Indigenous communities is still needed to catalyse conservation actions. Although little acknowledged in the conservation literature, this is often highly dependent on the personality and charisma of people within the Indigenous community, their ability to lead, tenacity to navigate bureaucratic and other challenges, capacity and knowledge of salient issues, astuteness in discerning possibilities to forward their agenda in politically amenable ways, having or being in the right situation that best-suits their skills, and importantly, be respected and listened to by the rest of the community (S. Nepstad & Bob, 2006). For example, Berta Cáceres, an Indigenous Lenca woman from Honduras, was an environmental defender and Indigenous rights campaigner who succeeded in preventing the Agua Zarca dam from being constructed on the sacred Gualcarque river in Lenca territory. Yet, for her leadership and activism, she was ordered to be murdered by the Honduran company executives planning the dam construction, and died in 2016. Although not necessarily an example of Indigenous-led conservation per se, nor one with a happy ending, Berta Cáceres embodied the traits necessary for Indigenous leadership for a positive environmental outcome, and her tragic murder highlights the severe challenges that remain to achieve Indigenous-led conservation.

A salient point regarding the success of Indigenous leadership is often recognition from State governments, having the legal means to obtain land tenure and to assert their rights. As briefly discussed in Section 1.2.2, the Indonesian government made it constitutionally possible for Indigenous communities to claim Customary Forest tenure in 2016. This was a big step in enabling Indigenous-led conservation since recognition of Indigenous peoples without accompanying rights to their land (such as in Nepal) greatly constrains their ability to take ownership, implement changes, and lead conservation which is inevitably tied to land. Land tenure, in particular land tenure security, is often associated with positive environmental and social outcomes (Section 2.4.2). Yet despite its central role in enabling Indigenous-led conservation, the crux of the issue is often State governments' denial of Indigenous land rights. In Kenya, the Indigenous Ogiek peoples from the Mau Forest Complex were evicted from their ancestral lands in October 2009 by the Kenya Forestry Service. The African Court on Human and Peoples' Rights issued a landmark ruling in their favour in May 2017, determining that the Kenyan government had discriminated against and violated the Ogieks' rights to their land, livelihood, and spiritual and cultural practices (Claridge, 2018), with a further judgement in June 2022 for the Kenyan authorities to compensate the Ogieks and grant them collective title to their ancestral lands. Though it remains to be seen how long the Kenyan government will take to implement the judgement, international human rights courts (the African Court on Human and Peoples' Rights, Inter-American Court of Human Rights, though the Asia-Pacific region is still lacking one) are likely a major recourse to obtaining Indigenous land rights for Indigenous peoples.

5.5. Conclusions and moving forward

Tropical forests are under increasing pressure (Edwards et al., 2019; Malhi et al., 2014); industrial agriculture, mining, and large-scale infrastructure driven by affluence and over-consumption continue to expand in footprint (Johnson et al., 2020; Wiedmann et al., 2020), while development banks that finance such development lack biodiversity safeguards and fail to consider their future impacts (Narain et al., 2023). Urban land expansion is also expected to contribute to habitat loss for many species, particularly in tropical regions of sub-Saharan Africa, Central and South America, and Southeast Asia (Simkin et al., 2022).

My thesis applies big data and robust statistical methods to find that Indigenous lands are important for reducing tropical deforestation and degradation and protecting vital habitats for biodiversity, contributing to conservation outcomes. Tackling the twin biodiversity and climate crisis and their impacts on human society will require more effective protection of remaining intact and functional species-rich and carbon-rich ecosystems, underpinned by political, economic, and social institutions that support more equitable and participatory development and access to good quality of life from local to global levels (Pörtner et al., 2023). The post-2020 Kunming-Montreal Global Biodiversity Framework provides the opportunity for transformative governance in area-based conservation. Yet going beyond securing a good quality of life for individuals, groups, and societies, a transformative approach to conservation will have to address the wider and deep-rooted drivers of the crises by shifting away from colonial and neoliberal conservation ideologies which reinforce the human-nature dichotomy and subsequent resulting inequalities (Büscher & Fletcher, 2019; Kashwan et al., 2021).

Following the wisdom of the well-used quote that we cannot solve problems using the same thinking we used to create them, to resolve the biodiversity and climate crisis, we have to adopt different tools and thinking. Colonial conservation relies on techno-fixes and top-down managerial approaches to area-based conservation, which, while effective in reducing habitat loss, has not managed to stem the overall decline and has often committed much injustice. New approaches that work with Indigenous peoples as equal partners and address the varied spatial and temporal scales and telecoupled drivers of the biodiversity and climate crises are needed (Carmenta et al., 2023). My thesis supports the implementation of broader policies for greater rights recognition of Indigenous peoples and moving towards more equitable and Indigenous-led approaches to conservation. Without Indigenous peoples as leaders and allies and their biocultural knowledge and tenacity to resist the Capitalocene (Moore, 2016), it is unlikely that humanity will be able to maintain a harmonious and liveable planet for humans and non-human species.

Bibliography

Abman, R., & Lundberg, C. (2020). Does Free Trade Increase Deforestation? The Effects of Regional Trade Agreements. *Journal of the Association of Environmental and Resource Economists*, *7*(1), 35–72. https://doi.org/10.1086/705787

Adams, W. M., & Hutton, J. (2007). People, parks and poverty: Political ecology and biodiversity conservation. *Conservation and Society*, *5*(2), 147–183.

Adams, W. M., & Mulligan, M. (Eds.). (2003). *Decolonizing nature: Strategies for conservation in a post-colonial era*. Earthscan Publications.

Alves-Pinto, H. N., Cordeiro, C. L. O., Geldmann, J., Jonas, H. D., Gaiarsa, M. P., Balmford, A., Watson, J. E. M., Latawiec, A. E., & Strassburg, B. (2022). The role of different governance regimes in reducing native vegetation conversion and promoting regrowth in the Brazilian Amazon. *Biological Conservation*, *267*, 109473. https://doi.org/10.1016/j.biocon.2022.109473

Amatulli, G., Domisch, S., Tuanmu, M.-N., Parmentier, B., Ranipeta, A., Malczyk, J., & Jetz, W. (2018). A suite of global, cross-scale topographic variables for environmental and biodiversity modeling. *Scientific Data*, *5*(1), 180040. https://doi.org/10.1038/sdata.2018.40

Anau, N., Hakim, A., Lekson, A. S., & Setyowati, E. (2019). Local Wisdom Practices of Dayak Indigenous People in the Management of Tana’ Ulen in the Kayan Mentarang National Park of Malinau Regency, North Kalimantan Province, Indonesia. *Russian Journal of Agricultural and Socio-Economic Sciences*, *91*(7), 156–167. https://doi.org/10.18551/rjoas.2019-07.16

Andam, K. S., Ferraro, P. J., Pfaff, A., Sanchez-Azofeifa, G. A., & Robalino, J. A. (2008). Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences*, *105*(42), 16089–16094. https://doi.org/10.1073/pnas.0800437105

Anongos, A., Whitmore, A., & Tebtebba (Organization) (Eds.). (2012). *Pitfalls & pipelines: Indigenous peoples and extractive industries*. Tebtebba Foundation.

Archer, L. J., Müller, H. S., Jones, L. P., Ma, H., Gleave, R. A., Da Silva Cerqueira, A., McMurdo Hamilton, T., & Shennan-Farpón, Y. (2022). Towards fairer conservation: Perspectives and ideas from early-career researchers. *People and Nature*, *4*(3), 612–626. https://doi.org/10.1002/pan3.10309

Arizona, Y., Wicaksono, M. T., & Vel, J. (2019). The Role of Indigeneity NGOs in the Legal Recognition of Adat Communities and Customary Forests in Indonesia. *The Asia Pacific Journal of Anthropology*, *20*(5), 487–506. https://doi.org/10.1080/14442213.2019.1670241

Armenteras, D., Rodríguez, N., & Retana, J. (2009). Are conservation strategies effective in avoiding the deforestation of the Colombian Guyana Shield? *Biological Conservation, 142*(7), 1411–1419. https://doi.org/10.1016/j.biocon.2009.02.002

Armitage, D., De Loë, R., & Plummer, R. (2012). Environmental governance and its implications for conservation practice: Environmental governance. *Conservation Letters*, *5*(4), 245–255. https://doi.org/10.1111/j.1755-263X.2012.00238.x

Armitage, D., Mbatha, P., Muhl, E., Rice, W., & Sowman, M. (2020). Governance principles for community-centered conservation in the post-2020 global biodiversity framework. *Conservation Science and Practice*, *2*(2). https://doi.org/10.1111/csp2.160

Asante, E. A., Ababio, S., & Boadu, K. B. (2017). The Use of Indigenous Cultural Practices by the Ashantis for the Conservation of Forests in Ghana. *SAGE Open*, *7*(1), 215824401668761. https://doi.org/10.1177/2158244016687611

Asia Indigenous Peoples Pact, Coalition for Tenure Justice in Indonesia, & Rights and Resources Initiative. (2020). *Under the Cover of COVID: New Laws in Asia Favor Business at the Cost of Indigenous Peoples' and Local Communities' Land and Territorial Rights*. Rights and Resources Initiative.

Asia Indigenous Peoples Pact, Forest Peoples Programme, & Deutsche Gesellschaft fur Internationale Zusammenarbeit GmbH. (2020). *Report of the Thematic Workshop on Human Rights as enabling condition in the post-2020 Global Biodiversity Framework* (p. 31). Asia Indigenous peoples Pact (AIPP).

Atleo, E. R. (2012). *Principles of tsawalk: An indigenous approach to global crisis*. UBC Press.

Ban, N. C., Frid, A., Reid, M., Edgar, B., Shaw, D., & Siwallace, P. (2018). Incorporate Indigenous perspectives for impactful research and effective management. *Nature Ecology & Evolution*, *2*(11), 1680–1683. https://doi.org/10.1038/s41559-018-0706-0

Baragwanath, K., & Bayi, E. (2020). Collective property rights reduce deforestation in the Brazilian Amazon. *Proceedings of the National Academy of Sciences*, *117*(34), 20495–20502. https://doi.org/10.1073/pnas.1917874117

Barlow, J., Gardner, T. A., Araujo, I. S., Ávila-Pires, T. C., Bonaldo, A. B., Costa, J. E., Esposito, M. C., Ferreira, L. V., Hawes, J., Hernandez, M. I. M., Hoogmoed, M. S., Leite, R. N., Lo-Man-Hung, N. F., Malcolm, J. R., Martins, M. B., Mestre, L. A. M., Miranda-Santos, R., Nunes-

Gutjahr, A. L., Overal, W. L., … Peres, C. A. (2007). Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences*, *104*(47), 18555–18560. https://doi.org/10.1073/pnas.0703333104

Barlow, J., Lennox, G. D., Ferreira, J., Berenguer, E., Lees, A. C., Nally, R. M., Thomson, J. R., Ferraz, S. F. de B., Louzada, J., Oliveira, V. H. F., Parry, L., Ribeiro de Castro Solar, R., Vieira, I. C. G., Aragão, L. E. O. C., Begotti, R. A., Braga, R. F., Cardoso, T. M., Jr, R. C. de O., Souza Jr, C. M., … Gardner, T. A. (2016). Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. *Nature*, *535*(7610), 144–147. https://doi.org/10.1038/nature18326

Bebbington, A. J., Chicchon, A., Cuba, N., Greenspan, E., Hecht, S., Bebbington, D. H., Kandel, S., Osborne, T., Ray, R., Rogan, J., & Sauls, L. (2020). Priorities for governing large-scale infrastructure in the tropics. *Proceedings of the National Academy of Sciences*, *117*(36), 21829–21833. https://doi.org/10.1073/pnas.2015636117

Bebbington, A. J., Humphreys Bebbington, D., Sauls, L. A., Rogan, J., Agrawal, S., Gamboa, C., Imhof, A., Johnson, K., Rosa, H., Royo, A., Toumbourou, T., & Verdum, R. (2018). Resource extraction and infrastructure threaten forest cover and community rights. *Proceedings of the National Academy of Sciences*, *115*(52), 13164–13173. https://doi.org/10.1073/pnas.1812505115

Benítez-López, A., Santini, L., Schipper, A. M., Busana, M., & Huijbregts, M. A. J. (2019). Intact but empty forests? Patterns of hunting-induced mammal defaunation in the tropics. *PLOS Biology*, *17*(5), e3000247. https://doi.org/10.1371/journal.pbio.3000247

Berkes, F. (2017). *Sacred Ecology* (4th ed.). Routledge. https://doi.org/10.4324/9781315114644

BirdLife International. (2020). *BirdLife Data Zone* [Computer software]. http://datazone.birdlife.org/species/requestdis

Bivand, R., Keitt, T., & Rowlingson, B. (2020). *Rgdal: Bindings for the 'Geospatial' Data Abstraction Library. R package version 1.4-8* [Computer software]. https://CRAN.R-project.org/package=rgdal

Bivand, R., & Rundel, C. (2020). *Rgeos: Interface to Geometry Engine—Open Source ('GEOS'). R package version 0.5-2* [Computer software]. https://CRAN.R-project.org/package=rgeos

Blackman, A., Corral, L., Lima, E. S., & Asner, G. P. (2017). Titling indigenous communities protects forests in the Peruvian Amazon. *Proceedings of the National Academy of Sciences*, *114*(16), 4123–4128. https://doi.org/10.1073/pnas.1603290114

Blackman, A., & Veit, P. (2018). Titled Amazon Indigenous Communities Cut Forest Carbon Emissions. *Ecological Economics*, *153*, 56–67. https://doi.org/10.1016/j.ecolecon.2018.06.016

Bland, L. M., Collen, B., Orme, C. D. L., & Bielby, J. (2015). Predicting the conservation status of data-deficient species: Predicting Extinction Risk. *Conservation Biology*, *29*(1), 250–259. https://doi.org/10.1111/cobi.12372

Blankespoor, B., Dasgupta, S., & Wheeler, D. (2017). Protected areas and deforestation: New results from high-resolution panel data. *Natural Resources Forum*, *41*(1), 55–68. https://doi.org/10.1111/1477-8947.12118

Bobo, K. S., Kamgaing, T. O. W., Ntumwel, B. C., Kagalang, D., Kengne, P. N. J., Ndengue, S. M. L., Badjeck, M. M. N., & Aghomo, F. F. M. (2014). Species richness, spatial distributions and densities of large and medium-sized mammals in the northern periphery of Boumba-Bek National Park, Southeastern, Cameroon. *African Study Monographs*, *49*, 91–114.

Borgelt, J., Dorber, M., Høiberg, M. A., & Verones, F. (2022). More than half of data deficient species predicted to be threatened by extinction. *Communications Biology*, *5*(1), 679. https://doi.org/10.1038/s42003-022-03638-9

Borrini-Feyerabend, G., Dudley, N., Jaegar, T., Lassen, B., Broome, N. P., Phillips, A., & Sandwith, T. (2013). *Governance of Protected Areas: From understanding to action* (20; Best Practice Protected Area Guidelines, p. 126). IUCN.

Boyce, P., Bhattacharyya, J., & Linklater, W. (2022). The need for formal reflexivity in conservation science. *Conservation Biology*, *36*(2). https://doi.org/10.1111/cobi.13840

Brandon, K., Redford, K. H., & Sanderson, S. E. (Eds.). (1998). *Parks in Peril: People, Politics, and Protected Areas*. Island Press.

Brittain, S., Ibbett, H., Lange, E., Dorward, L., Hoyte, S., Marino, A., Milner-Gulland, E. J., Newth, J., Rakotonarivo, S., Veríssimo, D., & Lewis, J. (2020). Ethical considerations when conservation research involves people. *Conservation Biology*, *34*(4), 925–933. https://doi.org/10.1111/cobi.13464

Brockington, D., & Igoe, J. (2006). Eviction for Conservation: A Global Overview. *Conservation and Society*, *4*(3), 424–470.

Brondízio, E. S., Aumeeruddy-Thomas, Y., Bates, P., Carino, J., Fernández-Llamazares, Á., Ferrari, M. F., Galvin, K., Reyes-García, V., McElwee, P., Molnár, Z., Samakov, A., & Shrestha, U. B. (2021). Locally Based, Regionally Manifested, and Globally Relevant: Indigenous and

Local Knowledge, Values, and Practices for Nature. *Annual Review of Environment and Resources*, *46*(1), 481–509. https://doi.org/10.1146/annurev-environ-012220-012127

Brooks, T. M., Akçakaya, H. R., Burgess, N. D., Butchart, S. H. M., Hilton-Taylor, C., Hoffmann, M., Juffe-Bignoli, D., Kingston, N., MacSharry, B., Parr, M., Perianin, L., Regan, E. C., Rodrigues, A. S. L., Rondinini, C., Shennan-Farpon, Y., & Young, B. E. (2016). Analysing biodiversity and conservation knowledge products to support regional environmental assessments. *Scientific Data*, *3*(1), 160007. https://doi.org/10.1038/sdata.2016.7

Brooks, T. M., Bakarr, M. I., Boucher, T., Da Fonseca, G. A. B., Hilton-Taylor, C., Hoekstra, J. M., Moritz, T., Olivieri, S., Parrish, J., Pressey, R. L., Rodrigues, A. S. L., Sechrest, W., Stattersfield, A., Strahm, W., & Stuart, S. N. (2004). Coverage Provided by the Global Protected-Area System: Is It Enough? *BioScience*, *54*(12), 1081. https://doi.org/10.1641/0006-3568(2004)054[1081:CPBTGP]2.0.CO;2

Brooks, T. M., Pimm, S. L., Akçakaya, H. R., Buchanan, G. M., Butchart, S. H. M., Foden, W., Hilton-Taylor, C., Hoffmann, M., Jenkins, C. N., Joppa, L., Li, B. V., Menon, V., Ocampo-Peñuela, N., & Rondinini, C. (2019). Measuring Terrestrial Area of Habitat (AOH) and Its Utility for the IUCN Red List. *Trends in Ecology & Evolution*, *34*(11), 977–986. https://doi.org/10.1016/j.tree.2019.06.009

Bruner, A. G., Gullison, R. E., Rice, R. E., & Da Fonseca, G. A. B. (2001). Effectiveness of Parks in Protecting Tropical Biodiversity. *Science*, *291*(5501), 125–128. https://doi.org/10.1126/science.291.5501.125

Bryan, J. (2019). For Nicaragua's Indigenous Communities, Land Rights in Name Only: Delineating boundaries among Indigenous and Black communities in eastern Nicaragua was supposed to guaranteed their land rights. Instead, it did the opposite. *NACLA Report on the Americas*, *51*(1), 55–64. https://doi.org/10.1080/10714839.2019.1593692

Burgess, R., Hansen, M., Olken, B. A., Potapov, P., & Sieber, S. (2012). The Political Economy of Deforestation in the Tropics. *The Quarterly Journal of Economics*, *127*(4), 1707–1754. https://doi.org/10.1093/qje/qjs034

Büscher, B., & Fletcher, R. (2019). Towards Convivial Conservation. *Conservation and Society*, *17*(3), 283. https://doi.org/10.4103/cs.cs_19_75

Butchart, S. H. M., & Bird, J. P. (2010). Data Deficient birds on the IUCN Red List: What don't we know and why does it matter? *Biological Conservation*, *143*(1), 239–247. https://doi.org/10.1016/j.biocon.2009.10.008

Carmenta, R., Barlow, J., Bastos Lima, M. G., Berenguer, E., Choiruzzad, S., Estrada-Carmona, N., França, F., Kallis, G., Killick, E., Lees, A., Martin, A., Pascual, U., Pettorelli, N., Reed, J., Rodriguez, I., Steward, A. M., Sunderland, T., Vira, B., Zaehringer, J. G., & Hicks, C. (2023). Connected Conservation: Rethinking conservation for a telecoupled world. *Biological Conservation*, *282*, 110047. https://doi.org/10.1016/j.biocon.2023.110047

Ceballos, G., Ehrlich, P. R., Barnosky, A. D., García, A., Pringle, R. M., & Palmer, T. M. (2015). Accelerated modern human–induced species losses: Entering the sixth mass extinction. *Science Advances*, *1*(5), e1400253. https://doi.org/10.1126/sciadv.1400253

Ceballos, G., Ehrlich, P. R., & Dirzo, R. (2017). Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. *Proceedings of the National Academy of Sciences*, *114*(30). https://doi.org/10.1073/pnas.1704949114

Ceddia, M. G. (2020). The super-rich and cropland expansion via direct investments in agriculture. *Nature Sustainability*, *3*(4), 312–318. https://doi.org/10.1038/s41893-020-0480-2

Ceddia, M. G., Gunter, U., & Corriveau-Bourque, A. (2015). Land tenure and agricultural expansion in Latin America: The role of Indigenous Peoples' and local communities' forest rights. *Global Environmental Change*, *35*, 316–322. https://doi.org/10.1016/j.gloenvcha.2015.09.010

Center for International Earth Science Information Network - CIESIN - Columbia University, & Information Technology Outreach Services - ITOS - University of Georgia. (2013). *Global Roads Open Access Data Set (gROADSv1), Version 1 (1980-2010)* [Computer software]. NASA Socioeconomic Data and Applications Center (SEDAC). https://doi.org/10.7927/H4VD6WCT

Chapela, F. (2005). Chapter 5: Indigenous Community Forest Management in the Sierra Juárez, Oaxaca. In D. B. Bray, L. Merino-Pérez, & D. Barry (Eds.), *The Community Forests of Mexico* (pp. 89–110). University of Texas Press. https://doi.org/10.7560/706378-007

Claridge, L. (2018). Litigation as a Tool for Community Empowerment: The Case of Kenya's Ogiek. *Erasmus Law Review*, *11*(1), 57–66. https://doi.org/10.5553/ELR.000095

Coad, L., Watson, J. E., Geldmann, J., Burgess, N. D., Leverington, F., Hockings, M., Knights, K., & Di Marco, M. (2019). Widespread shortfalls in protected area resourcing undermine efforts to conserve biodiversity. *Frontiers in Ecology and the Environment*, *17*(5), 259–264. https://doi.org/10.1002/fee.2042

Coetzee, B. W. T., Gaston, K. J., & Chown, S. L. (2014). Local Scale Comparisons of Biodiversity as a Test for Global Protected Area Ecological Performance: A Meta-Analysis. *PLoS ONE*, *9*(8), e105824. https://doi.org/10.1371/journal.pone.0105824

Colchester, M. (2004). Conservation policy and indigenous peoples. *Environmental Science & Policy*, *7*(3), 145–153. https://doi.org/10.1016/j.envsci.2004.02.004

Colding, J., & Folke, C. (2001). Social Taboos: 'Invisible' systems of local resource management and biological conservation. *Ecological Applications*, *11*(2), 584–600.

Conigliani, C., Cuffaro, N., & D'Agostino, G. (2018). Large-scale land investments and forests in Africa. *Land Use Policy*, *75*, 651–660. https://doi.org/10.1016/j.landusepol.2018.02.005

Conservation Initiative on Human Rights. (2014). *Human rights in conservation: Progress since Durban*. https://www.ohchr.org/sites/default/files/Documents/Issues/Environment/ImplementationReport/IUCN5.pdf

Convention on Biological Diversity. (2010). *Aichi Biodiversity Targets*. https://www.cbd.int/aichi-targets/

Convention on Biological Diversity. (2020). *Zero Draft of the Post-2020 Global Biodiversity Framework. CBD/WG2020/2/3*. Convention on Biological Diversity. https://www.cbd.int/doc/c/efb0/1f84/a892b98d2982a829962b6371/wg2020-02-03-en.pdf

Convention on Biological Diversity. (2022). *Kunming-Montreal Global Biodiversity Framework* (CBD/COP/15/L25). UNEP Convention on Biological Diversity. https://www.cbd.int/doc/c/e6d3/cd1d/daf663719a03902a9b116c34/cop-15-l-25-en.pdf

Corrigan, C., Bingham, H., Shi, Y., Lewis, E., Chauvenet, A., & Kingston, N. (2018). Quantifying the contribution to biodiversity conservation of protected areas governed by indigenous peoples and local communities. *Biological Conservation*, *227*, 403–412. https://doi.org/10.1016/j.biocon.2018.09.007

Curtis, P. G., Slay, C. M., Harris, N. L., Tyukavina, A., & Hansen, M. C. (2018). Classifying drivers of global forest loss. *Science*, *361*(6407), 1108–1111. https://doi.org/10.1126/science.aau3445

Davis, K. F., Yu, K., Rulli, M. C., Pichdara, L., & D'Odorico, P. (2015). Accelerated deforestation driven by large-scale land acquisitions in Cambodia. *Nature Geoscience*, *8*(10), 772–775. https://doi.org/10.1038/ngeo2540

Davis, S. H., & Wali, A. (1994). Indigenous land tenure and tropical forest management in Latin America. *Ambio*, *23*(8), 485–490.

Dawson, N. M., Coolsaet, B., Sterling, E. J., Loveridge, R., Gross-Camp, N. D., Wongbusarakum, S., Sangha, K. K., Scherl, L. M., Phan, H. P., Zafra-Calvo, N., Lavey, W. G., Byakagaba, P., Idrobo, C. J., Chenet, A., Bennett, N. J., Mansourian, S., & Rosado-May, F. J. (2021). The role of Indigenous peoples and local communities in effective and equitable conservation. *Ecology and Society*, *26*(3), art19. https://doi.org/10.5751/ES-12625-260319

Di Marco, M., Watson, J. E. M., Possingham, H. P., & Venter, O. (2017). Limitations and trade-offs in the use of species distribution maps for protected area planning. *Journal of Applied Ecology*, *54*(2), 402–411. https://doi.org/10.1111/1365-2664.12771

Dinerstein, E., Olson, D., Joshi, A., Vynne, C., Burgess, N. D., Wikramanayake, E., Hahn, N., Palminteri, S., Hedao, P., Noss, R., Hansen, M., Locke, H., Ellis, E. C., Jones, B., Barber, C. V., Hayes, R., Kormos, C., Martin, V., Crist, E., … Saleem, M. (2017). An Ecoregion-Based Approach to Protecting Half the Terrestrial Realm. *BioScience*, *67*(6), 534–545. https://doi.org/10.1093/biosci/bix014

Dinerstein, E., Vynne, C., Sala, E., Joshi, A. R., Fernando, S., Lovejoy, T. E., Mayorga, J., Olson, D., Asner, G. P., Baillie, J. E. M., Burgess, N. D., Burkart, K., Noss, R. F., Zhang, Y. P., Baccini, A., Birch, T., Hahn, N., Joppa, L. N., & Wikramanayake, E. (2019). A Global Deal For Nature: Guiding principles, milestones, and targets. *Science Advances*, *5*(4), eaaw2869. https://doi.org/10.1126/sciadv.aaw2869

Dirzo, R., Young, H. S., Galetti, M., Ceballos, G., Isaac, N. J. B., & Collen, B. (2014). Defaunation in the Anthropocene. *Science*, *345*(6195), 401–406. https://doi.org/10.1126/science.1251817

Domínguez, L., & Luoma, C. (2020). Decolonising Conservation Policy: How Colonial Land and Conservation Ideologies Persist and Perpetuate Indigenous Injustices at the Expense of the Environment. *Land*, *9*(3), 65. https://doi.org/10.3390/land9030065

dos Santos, J. G., Malhado, A. C. M., Ladle, R. J., Correia, R. A., & Costa, M. H. (2015). Geographic trends and information deficits in Amazonian conservation research. *Biodiversity and Conservation*, *24*(11), 2853–2863. https://doi.org/10.1007/s10531-015-0981-x

Dressler, W. (2021). Quarantining activism: COVID-19, frontline defenders and intensifying extractivism in the Philippines. *Journal of Political Ecology*, *28*(1). https://doi.org/10.2458/jpe.2955

Dubertret, F., & Alden Wily, L. (2015). *Percent of Indigenous and Community Lands. Data file from Landmark: The GLobal Platform of Indigenous and Community Lands* [Computer software]. www.landmarkmap.org

Dudley, N. (Ed.). (2008). *Guidelines for Applying Protected Area Management Categories*. IUCN. https://portals.iucn.org/library/sites/library/files/documents/pag-021.pdf

Dudley, N., Belokurov, A., Borodin, O., Higgins-Zogib, L., Hockings, M., Lacerda, L., & Stolton, S. (2004). *Are protected areas working?: An analysis of forest protected areas by WWF*. WWF International. https://wwfeu.awsassets.panda.org/downloads/areprotectedareasworking.pdf

Dudley, N., Jonas, H., Nelson, F., Parrish, J., Pyhälä, A., Stolton, S., & Watson, J. E. M. (2018). The essential role of other effective area-based conservation measures in achieving big bold conservation targets. *Global Ecology and Conservation*, *15*, e00424. https://doi.org/10.1016/j.gecco.2018.e00424

Durán, A. P., Green, J. M. H., West, C. D., Visconti, P., Burgess, N. D., Virah-Sawmy, M., & Balmford, A. (2020). A practical approach to measuring the biodiversity impacts of land conversion. *Methods in Ecology and Evolution*, *11*(8), 910–921. https://doi.org/10.1111/2041-210X.13427

Duran-Izquierdo, M., & Olivero-Verbel, J. (2021). Vulnerability assessment of Sierra Nevada de Santa Marta, Colombia: World's most irreplaceable nature reserve. *Global Ecology and Conservation*, *28*, e01592. https://doi.org/10.1016/j.gecco.2021.e01592

Dutta, A., Allan, J., Worsdell, T., Duffy, R., Kumar, K., Rai, N., Fischer, H. W., Shimray, G., & Sherpa, P. D. (2021). Response to A "Global Safety Net" to reverse biodiversity loss and stabilize Earth's climate. *Science Advances*, *6*(36). https://advances.sciencemag.org/content/6/36/eabb2824/tab-e-letters

ECLAC. (2014). *Guaranteeing Indigenous people's rights in Latin America: Progress in the past decade and remaining challenges*. United Nations Economic Commission for Latin America and the Caribbean. https://repositorio.cepal.org/bitstream/handle/11362/37051/4/S1420782_en.pdf

Edwards, D. P., Socolar, J. B., Mills, S. C., Burivalova, Z., Koh, L. P., & Wilcove, D. S. (2019). Conservation of Tropical Forests in the Anthropocene. *Current Biology*, *29*(19), R1008–R1020. https://doi.org/10.1016/j.cub.2019.08.026

Edwards, D. P., Tobias, J. A., Sheil, D., Meijaard, E., & Laurance, W. F. (2014). Maintaining ecosystem function and services in logged tropical forests. *Trends in Ecology & Evolution*, *29*(9), 511–520. https://doi.org/10.1016/j.tree.2014.07.003

Ellis, E. C., Gauthier, N., Klein Goldewijk, K., Bliege Bird, R., Boivin, N., Díaz, S., Fuller, D. Q., Gill, J. L., Kaplan, J. O., Kingston, N., Locke, H., McMichael, C. N. H., Ranco, D., Rick, T.

C., Shaw, M. R., Stephens, L., Svenning, J.-C., & Watson, J. E. M. (2021). People have shaped most of terrestrial nature for at least 12,000 years. *Proceedings of the National Academy of Sciences*, *118*(17), e2023483118. https://doi.org/10.1073/pnas.2023483118

Ens, E., Reyes-García, V., Asselin, H., Hsu, M., Reimerson, E., Reihana, K., Sithole, B., Shen, X., Cavanagh, V., & Adams, M. (2021). Recognition of Indigenous Ecological Knowledge Systems in Conservation and Their Role to Narrow the Knowledge-Implementation Gap. In C. C. Ferreira & C. F. C. Klütsch (Eds.), *Closing the Knowledge-Implementation Gap in Conservation Science* (Vol. 4, pp. 109–139). Springer International Publishing. https://doi.org/10.1007/978-3-030-81085-6_5

Environmental Systems Research Institute. (2019). *ArcGIS desktop* (10.4.7) [Computer software]. Esri. https://desktop.arcgis.com/en/arcmap

Escobar, A. (2011). *Encountering Development: The Making and Unmaking of the Third World*. Princeton University Press.

Estrada, A., Garber, P. A., Gouveia, S., Fernández-Llamazares, Á., Ascensão, F., Fuentes, A., Garnett, S. T., Shaffer, C., Bicca-Marques, J., Fa, J. E., Hockings, K., Shanee, S., Johnson, S., Shepard, G. H., Shanee, N., Golden, C. D., Cárdenas-Navarrete, A., Levey, D. R., Boonratana, R., … Volampeno, S. (2022). Global importance of Indigenous Peoples, their lands, and knowledge systems for saving the world's primates from extinction. *Science Advances*, *8*(32), eabn2927. https://doi.org/10.1126/sciadv.abn2927

European Parliament. Directorate General for External Policies of the Union. (2014). *Indigenous peoples, extractive industries and human rights.* Publications Office. https://data.europa.eu/doi/10.2861/69341

Ewers, R. M., Boyle, M. J. W., Gleave, R. A., Plowman, N. S., Benedick, S., Bernard, H., Bishop, T. R., Bakhtiar, E. Y., Chey, V. K., Chung, A. Y. C., Davies, R. G., Edwards, D. P., Eggleton, P., Fayle, T. M., Hardwick, S. R., Homathevi, R., Kitching, R. L., Khoo, M. S., Luke, S. H., … Turner, E. C. (2015). Logging cuts the functional importance of invertebrates in tropical rainforest. *Nature Communications*, *6*(1), 6836. https://doi.org/10.1038/ncomms7836

Fa, J. E., Watson, J. E., Leiper, I., Potapov, P., Evans, T. D., Burgess, N. D., Molnár, Z., Fernández-Llamazares, Á., Duncan, T., Wang, S., Austin, B. J., Jonas, H., Robinson, C. J., Malmer, P., Zander, K. K., Jackson, M. V., Ellis, E., Brondizio, E. S., & Garnett, S. T. (2020). Importance of Indigenous Peoples' lands for the conservation of Intact Forest Landscapes. *Frontiers in Ecology and the Environment*, *18*(3), 135–140. https://doi.org/10.1002/fee.2148

Fernández-Llamazares, Á., López-Baucells, A., Velazco, P. M., Gyawali, A., Rocha, R., Terraube, J., & Cabeza, M. (2021). The importance of Indigenous Territories for conserving bat diversity across the Amazon biome. *Perspectives in Ecology and Conservation*, *19*(1), 10–20. https://doi.org/10.1016/j.pecon.2020.11.001

Ferraro, P. J., & Pattanayak, S. K. (2006). Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments. *PLoS Biology*, *4*(4), e105. https://doi.org/10.1371/journal.pbio.0040105

Ficetola, G. F., Rondinini, C., Bonardi, A., Katariya, V., Padoa-Schioppa, E., & Angulo, A. (2014). An evaluation of the robustness of global amphibian range maps. *Journal of Biogeography*, *41*(2), 211–221. https://doi.org/10.1111/jbi.12206

Finer, M., Jenkins, C. N., Pimm, S. L., Keane, B., & Ross, C. (2008). Oil and Gas Projects in the Western Amazon: Threats to Wilderness, Biodiversity, and Indigenous Peoples. *PLoS ONE*, *3*(8), e2932. https://doi.org/10.1371/journal.pone.0002932

Fletcher, M.-S., Hamilton, R., Dressler, W., & Palmer, L. (2021). Indigenous knowledge and the shackles of wilderness. *Proceedings of the National Academy of Sciences*, *118*(40), e2022218118. https://doi.org/10.1073/pnas.2022218118

Fletcher, R., & Büscher, B. (2020). Conservation basic income: A non-market mechanism to support convivial conservation. *Biological Conservation*, *244*, 108520. https://doi.org/10.1016/j.biocon.2020.108520

Ford, J. D., King, N., Galappaththi, E. K., Pearce, T., McDowell, G., & Harper, S. L. (2020). The Resilience of Indigenous Peoples to Environmental Change. *One Earth*, *2*(6), 532–543. https://doi.org/10.1016/j.oneear.2020.05.014

Forest Peoples Programme, International Indigenous Forum on Biodiversity, Indigenous Women's Biodiversity Network, Centres of Distinction on Indigenous and Local Knowledge, & Secretariat of the Convention on Biological Diversity. (2020). *Local Biodiversity Outlooks 2: The contributions of indigenous peoples and local communities to the implementation of the Strategic Plan for Biodiversity 2011-2020 and to renewing nature and cultures. A complement to the fifth edition of the Global Biodiversity Outlooks* (p. 312). Forest Peoples Programme. www.localbiodiversityoutlooks.net

Forister, M. L., Pelton, E. M., & Black, S. H. (2019). Declines in insect abundance and diversity: We know enough to act now. *Conservation Science and Practice*, *1*(8). https://doi.org/10.1111/csp2.80

Fuller, C., Ondei, S., Brook, B. W., & Buettel, J. C. (2019). First, do no harm: A systematic review of deforestation spillovers from protected areas. *Global Ecology and Conservation*, *18*, e00591. https://doi.org/10.1016/j.gecco.2019.e00591

Gardner, C. J., Bicknell, J. E., Baldwin-Cantello, W., Struebig, M. J., & Davies, Z. G. (2019). Quantifying the impacts of defaunation on natural forest regeneration in a global meta-analysis. *Nature Communications*, *10*(1), 4590. https://doi.org/10.1038/s41467-019-12539-1

Garnett, S. T., Burgess, N. D., Fa, J. E., Fernández-Llamazares, Á., Molnár, Z., Robinson, C. J., Watson, J. E. M., Zander, K. K., Austin, B., Brondizio, E. S., Collier, N. F., Duncan, T., Ellis, E., Geyle, H., Jackson, M. V., Jonas, H., Malmer, P., McGowan, B., Sivongxay, A., & Leiper, I. (2018). A spatial overview of the global importance of Indigenous lands for conservation. *Nature Sustainability*, *1*(7), 369–374. https://doi.org/10.1038/s41893-018-0100-6

Gaveau, D. L. A., Epting, J., Lyne, O., Linkie, M., Kumara, I., Kanninen, M., & Leader-Williams, N. (2009). Evaluating whether protected areas reduce tropical deforestation in Sumatra. *Journal of Biogeography*, *36*(11), 2165–2175. https://doi.org/10.1111/j.1365-2699.2009.02147.x

Geldmann, J., Barnes, M., Coad, L., Craigie, I. D., Hockings, M., & Burgess, N. D. (2013). Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation*, *161*, 230–238. https://doi.org/10.1016/j.biocon.2013.02.018

Geldmann, J., Manica, A., Burgess, N. D., Coad, L., & Balmford, A. (2019). A global-level assessment of the effectiveness of protected areas at resisting anthropogenic pressures. *Proceedings of the National Academy of Sciences*, *116*(46), 23209–23215. https://doi.org/10.1073/pnas.1908221116

Gibson, L., Lee, T. M., Koh, L. P., Brook, B. W., Gardner, T. A., Barlow, J., Peres, C. A., Bradshaw, C. J. A., Laurance, W. F., Lovejoy, T. E., & Sodhi, N. S. (2011). Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature*, *478*(7369), 378–381. https://doi.org/10.1038/nature10425

Giljum, S., Maus, V., Kuschnig, N., Luckeneder, S., Tost, M., Sonter, L. J., & Bebbington, A. J. (2022). A pantropical assessment of deforestation caused by industrial mining. *Proceedings of the National Academy of Sciences*, *119*(38), e2118273119. https://doi.org/10.1073/pnas.2118273119

Global Administrative Areas. (2018). *GAD database of Global Administrative Areas, version 3.6* (3.6) [Computer software]. https://gadm.org

Global Forest Watch. (2015). *Tree cover 2000* (1.2) [Computer software].

Global Forest Watch. (2018). *Tree cover loss* (1.6) [Computer software]. www.globalforestwatch.org

Global Forest Watch. (2019). *Spatial Database of Planted Trees* (1.0) [Computer software]. https://data.globalforestwatch.org/documents/gfw::planted-forests/about

Global Land Analysis & Discovery, Department of Geographical Sciences, University of Maryland. (2013). *Global 2010 Tree Cover (30m)* [Computer software]. https://glad.umd.edu/dataset/global-2010-tree-cover-30-m

Golden Kroner, R. E., Barbier, E. B., Chassot, O., Chaudhary, S., Cordova Jr, L., Cruz-Trinidad, A., Cumming, T., Howard, J., Karibuhoye Said, C., Kun, Z., Ogena, A., Palla, F., Samayoa Valiente, R., Troëng, S., Valverde, A., Wijethunga, R., & Wong, M. (2021). COVID-era policies and economic recovery plans: Are governments building back better for protected and conserved areas? *PARKS*, *27*, 135–148. https://doi.org/10.2305/IUCN.CH.2021.PARKS-27-SIRGK.en

Golden Kroner, R. E., Qin, S., Cook, C. N., Krithivasan, R., Pack, S. M., Bonilla, O. D., Cort-Kansinally, K. A., Coutinho, B., Feng, M., Martínez Garcia, M. I., He, Y., Kennedy, C. J., Lebreton, C., Ledezma, J. C., Lovejoy, T. E., Luther, D. A., Parmanand, Y., Ruíz-Agudelo, C. A., Yerena, E., … Mascia, M. B. (2019). The uncertain future of protected lands and waters. *Science*, *364*(6443), 881–886. https://doi.org/10.1126/science.aau5525

González-del-Pliego, P., Freckleton, R. P., Edwards, D. P., Koo, M. S., Scheffers, B. R., Pyron, R. A., & Jetz, W. (2019). Phylogenetic and Trait-Based Prediction of Extinction Risk for Data-Deficient Amphibians. *Current Biology*, *29*(9), 1557-1563.e3. https://doi.org/10.1016/j.cub.2019.04.005

Grantham, H. S., Duncan, A., Evans, T. D., Jones, K. R., Beyer, H. L., Schuster, R., Walston, J., Ray, J. C., Robinson, J. G., Callow, M., Clements, T., Costa, H. M., DeGemmis, A., Elsen, P. R., Ervin, J., Franco, P., Goldman, E., Goetz, S., Hansen, A., … Watson, J. E. M. (2020). Anthropogenic modification of forests means only 40% of remaining forests have high ecosystem integrity. *Nature Communications*, *11*(1), 5978. https://doi.org/10.1038/s41467-020-19493-3

Grantham, H. S., Tibaldeschi, P., Izquierdo, P., Mo, K., Patterson, D. J., Rainey, H., Watson, J. E. M., & Jones, K. R. (2021). The Emerging Threat of Extractives Sector to Intact Forest Landscapes. *Frontiers in Forests and Global Change*, *4*, 692338. https://doi.org/10.3389/ffgc.2021.692338

Gray, C. L., Hill, S. L. L., Newbold, T., Hudson, L. N., Börger, L., Contu, S., Hoskins, A. J., Ferrier, S., Purvis, A., & Scharlemann, J. P. W. (2016). Local biodiversity is higher inside than outside terrestrial protected areas worldwide. *Nature Communications*, *7*(1), 12306. https://doi.org/10.1038/ncomms12306

Guerin, G. R., & Lowe, A. J. (2015). 'Sum of inverse range-sizes' (SIR), a biodiversity metric with many names and interpretations. *Biodiversity and Conservation*, *24*(11), 2877–2882. https://doi.org/10.1007/s10531-015-0977-6

Gurney, G. G., Adams, V. M., Álvarez-Romero, J. G., & Claudet, J. (2023). Area-based conservation: Taking stock and looking ahead. *One Earth*, *6*(2), 98–104. https://doi.org/10.1016/j.oneear.2023.01.012

Haenssgen, M. J., Lechner, A. M., Rakotonarivo, S., Leepreecha, P., Sakboon, M., Chu, T.-W., Auclair, E., & Vlaev, I. (2022). Implementation of the COP26 declaration to halt forest loss must safeguard and include Indigenous people. *Nature Ecology & Evolution*. https://doi.org/10.1038/s41559-021-01650-6

Haines, K. (2021). Oaxaca and global forest governance: Indigenous autonomy, local institutions, and forest outcomes in Southern Mexico. *Journal of Political Ecology*, *28*(1). https://doi.org/10.2458/jpe.2296

Hansen, A. J., Burns, P., Ervin, J., Goetz, S. J., Hansen, M., Venter, O., Watson, J. E. M., Jantz, P. A., Virnig, A. L. S., Barnett, K., Pillay, R., Atkinson, S., Supples, C., Rodríguez-Buritica, S., & Armenteras, D. (2020). A policy-driven framework for conserving the best of Earth's remaining moist tropical forests. *Nature Ecology & Evolution*, *4*(10), 1377–1384. https://doi.org/10.1038/s41559-020-1274-7

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O., & Townshend, J. R. G. (2013). High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, *342*(6160), 850–853. https://doi.org/10.1126/science.1244693

Hansen, M. C., Stehman, S. V., & Potapov, P. V. (2010). Quantification of global gross forest cover loss. *Proceedings of the National Academy of Sciences*, *107*(19), 8650–8655. https://doi.org/10.1073/pnas.0912668107

Hanson, J. O. (2020). *wdpar: Interface to the world database on protected areas* [Computer software]. https://CRAN.R-project.org/package=wdpar

Harvey, C. A., Gonzalez, J., & Somarriba, E. (2006). Dung Beetle and Terrestrial Mammal Diversity in Forests, Indigenous Agroforestry Systems and Plantain Monocultures in Talamanca, Costa Rica. *Biodiversity and Conservation*, *15*(2), 555–585. https://doi.org/10.1007/s10531-005-2088-2

Hayes, T. M., & Murtinho, F. (2008). Are indigenous forest reserves sustainable? An analysis of present and future land-use trends in Bosawas, Nicaragua. *International Journal of Sustainable Development & World Ecology*, *15*(6), 497–511. https://doi.org/10.1080/13504500809469845

Heckenberger, M. J., Christian Russell, J., Toney, J. R., & Schmidt, M. J. (2007). The legacy of cultural landscapes in the Brazilian Amazon: Implications for biodiversity. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *362*(1478), 197–208. https://doi.org/10.1098/rstb.2006.1979

Henders, S., Persson, U. M., & Kastner, T. (2015). Trading forests: Land-use change and carbon emissions embodied in production and exports of forest-risk commodities. *Environmental Research Letters*, *10*(12), 125012. https://doi.org/10.1088/1748-9326/10/12/125012

Herrera, D., Pfaff, A., & Robalino, J. (2019). Impacts of protected areas vary with the level of government: Comparing avoided deforestation across agencies in the Brazilian Amazon. *Proceedings of the National Academy of Sciences*, *116*(30), 14916–14925. https://doi.org/10.1073/pnas.1802877116

Hidayat, H., Yogaswara, H., Herawati, T., Blazey, P., Wyatt, S., & Howitt, R. (2018). Forests, law and customary rights in Indonesia: Implications of a decision of the Indonesian Constitutional Court in 2012. *Asia Pacific Viewpoint*, *59*(3), 293–308. https://doi.org/10.1111/apv.12207

Hijmans, R. J. (2020). *raster: Geographic data analysis and modeling* [Computer software]. https://CRAN.R-project.org/package=raster

Hill, R., Nates-Parra, G., Quezada-Euán, J. J. G., Buchori, D., LeBuhn, G., Maués, M. M., Pert, P. L., Kwapong, P. K., Saeed, S., Breslow, S. J., Carneiro da Cunha, M., Dicks, L. V., Galetto, L., Gikungu, M., Howlett, B. G., Imperatriz-Fonseca, V. L., O'B. Lyver, P., Martín-López, B., Oteros-Rozas, E., … Roué, M. (2019). Biocultural approaches to pollinator conservation. *Nature Sustainability*, *2*(3), 214–222. https://doi.org/10.1038/s41893-019-0244-z

Hill, S. L. L., Arnell, A., Maney, C., Butchart, S. H. M., Hilton-Taylor, C., Ciciarelli, C., Davis, C., Dinerstein, E., Purvis, A., & Burgess, N. D. (2019). Measuring Forest Biodiversity Status and Changes Globally. *Frontiers in Forests and Global Change*, *2*, 70. https://doi.org/10.3389/ffgc.2019.00070

Ho, D., Imai, K., King, G., & Stuart, E. (2018). *MatchIt: Nonparametric preprocessing for parametric causal inference* [Manual]. https://CRAN.R-project.org/package=MatchIt

Hoang, N. T., & Kanemoto, K. (2021). Mapping the deforestation footprint of nations reveals growing threat to tropical forests. *Nature Ecology & Evolution*, *5*(6), 845–853. https://doi.org/10.1038/s41559-021-01417-z

Hoang, N. T., Taherzadeh, O., Ohashi, H., Yonekura, Y., Nishijima, S., Yamabe, M., Matsui, T., Matsuda, H., Moran, D., & Kanemoto, K. (2023). Mapping potential conflicts between global agriculture and terrestrial conservation. *Proceedings of the National Academy of Sciences*, *120*(23), e2208376120. https://doi.org/10.1073/pnas.2208376120

Holmes, A. G. D. (2020). Researcher Positionality—A Consideration of Its Influence and Place in Qualitative Research—A New Researcher Guide. *Shanlax International Journal of Education*, *8*(4), 1–10. https://doi.org/10.34293/education.v8i4.3232

How Jin Aik, D., Ismail, M. H., Muharam, F. M., & Alias, M. A. (2021). Evaluating the impacts of land use/land cover changes across topography against land surface temperature in Cameron Highlands. *PLOS ONE*, *16*(5), e0252111. https://doi.org/10.1371/journal.pone.0252111

ICCA Consortium. (2021). *Territories of Life: 2021 Report* (p. 52). ICCA Consortium. https://report.territoriesoflife.org

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. (2019). *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (p. 56). IPBES secretariat. www.ipbes.net

International Indigenous Peoples’ Forum on Climate Change. (2022). *Statement of the International Indigenous Peoples Forum on Climate Change (IIPFCC) Joint Opening Plenary at COP27, Sharm El Sheikh, Egypt*. https://static1.squarespace.com/static/5627862ce4b07be93cfb9461/t/6368fe8b8f08b75317b4adf1/1667825291407/IIPFCC_Statement_6_Nov_2022.pdf

Indigenous and Tribal Peoples Convention (No. 169), 169 (1989). https://www.ilo.org/dyn/normlex/en/f?p=NORMLEXPUB:12100:0::NO::P12100_ILO_CODE:C169

Ismail, S. S., Abd Hamid, Z., Fakri, S. N. S. M., & Ab Aziz, L. H. (2021). Rural Tourism Development in Cameron Highlands: Indigenous Community’s Perception towards Socio-

Cultural and Economic Impacts. *International Journal of Academic Research in Business and Social Sciences*, *11*(13), Pages 206-221. https://doi.org/10.6007/IJARBSS/v11-i13/8516

IUCN. (1994). *Guidelines for Protected Area Management Categories.* CNPPA with the assistance of WCMC. IUCN, Gland, Switzerland and Cambridge, UK. https://portals.iucn.org/library/efiles/documents/1994-007-En.pdf

IUCN. (1996a). *Resolution 51: Indigenous Peoples, Mineral and Oil Extraction, Infrastructure and Development Works*. https://portals.iucn.org/library/node/44455

IUCN. (1996b). *Resolution 53: Indigenous Peoples and Protected Areas*. https://portals.iucn.org/library/node/44457

IUCN. (2019). *IUCN Standard on Indigenous Peoples*. IUCN. https://www.iucn.org/sites/default/files/2022-05/iucn-esms-standard-on-indigenous-people.pdf

IUCN. (2020). *The IUCN Red List of Threatened Species* (Version 2020-2) [Computer software]. https://www.iucnredlist.org/resources/spatial-data-download

IUCN. (2023). *The IUCN Red List of Threatened Species* (Version 2022-2) [dataset]. https://www.iucnredlist.org

Jackson, J. E., & Warren, K. B. (2005). Indigenous Movements In Latin America, 1992–2004: Controversies, Ironies, New Directions. *Annual Review of Anthropology*, *34*(1), 549–573. https://doi.org/10.1146/annurev.anthro.34.081804.120529

Jaureguiberry, P., Titeux, N., Wiemers, M., Bowler, D. E., Coscieme, L., Golden, A. S., Guerra, C. A., Jacob, U., Takahashi, Y., Settele, J., Díaz, S., Molnár, Z., & Purvis, A. (2022). The direct drivers of recent global anthropogenic biodiversity loss. *Science Advances*, *8*(45), eabm9982. https://doi.org/10.1126/sciadv.abm9982

Jetz, W., & Freckleton, R. P. (2015). Towards a general framework for predicting threat status of data-deficient species from phylogenetic, spatial and environmental information. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*(1662), 20140016. https://doi.org/10.1098/rstb.2014.0016

Johnson, C. J., Venter, O., Ray, J. C., & Watson, J. E. M. (2020). Growth-inducing infrastructure represents transformative yet ignored keystone environmental decisions. *Conservation Letters*, *13*(2). https://doi.org/10.1111/conl.12696

Jonas, H. D., Barbuto, V., Jonas, H. C., Kothari, A., & Nelson, F. (2014). New Steps of Change: Looking Beyond Protected Areas to Consider Other Effective Area-based Conservation

Measures. *PARKS*, *20.2*, 111–128. https://doi.org/10.2305/IUCN.CH.2014.PARKS-20-2.HDJ.en

Joppa, L. N., & Pfaff, A. (2009). High and Far: Biases in the Location of Protected Areas. *PLoS ONE*, *4*(12), e8273. https://doi.org/10.1371/journal.pone.0008273

Joppa, L. N., & Pfaff, A. (2011). Global protected area impacts. *Proceedings of the Royal Society B: Biological Sciences*, *278*(1712), 1633–1638. https://doi.org/10.1098/rspb.2010.1713

Jusys, T. (2018). Changing patterns in deforestation avoidance by different protection types in the Brazilian Amazon. *PLOS ONE*, *13*(4), e0195900. https://doi.org/10.1371/journal.pone.0195900

Kashwan, P., V. Duffy, R., Massé, F., Asiyanbi, A. P., & Marijnen, E. (2021). From Racialized Neocolonial Global Conservation to an Inclusive and Regenerative Conservation. *Environment: Science and Policy for Sustainable Development*, *63*(4), 4–19. https://doi.org/10.1080/00139157.2021.1924574

Kastner, T., Chaudhary, A., Gingrich, S., Marques, A., Persson, U. M., Bidoglio, G., Le Provost, G., & Schwarzmüller, F. (2021). Global agricultural trade and land system sustainability: Implications for ecosystem carbon storage, biodiversity, and human nutrition. *One Earth*, *4*(10), 1425–1443. https://doi.org/10.1016/j.oneear.2021.09.006

Kastner, T., Matej, S., Forrest, M., Gingrich, S., Haberl, H., Hickler, T., Krausmann, F., Lasslop, G., Niedertscheider, M., Plutzar, C., Schwarzmüller, F., Steinkamp, J., & Erb, K. (2022). Land use intensification increasingly drives the spatiotemporal patterns of the global human appropriation of net primary production in the last century. *Global Change Biology*, *28*(1), 307–322. https://doi.org/10.1111/gcb.15932

Kemf, E. (1993). *Indigenous peoples and protected areas: The law of Mother Earth*. Earthscan Publications Ltd.

Kingsbury, B. (1998). “Indigenous Peoples” in International Law: A Constructivist Approach to the Asian Controversy. *American Journal of International Law*, *92*(3), 414–457. https://doi.org/10.2307/2997916

Kothari, A., Camill, P., & Brown, J. (2013). Conservation as if People Also Mattered: Policy and Practice of Community-based Conservation. *Conservation and Society*, *11*(1), 1–15. https://doi.org/10.4103/0972-4923.110937

Larsen, P. B., Le Billon, P., Menton, M., Aylwin, J., Balsiger, J., Boyd, D., Forst, M., Lambrick, F., Santos, C., Storey, H., & Wilding, S. (2020). Understanding and responding to the environmental human rights defenders crisis: The case for conservation action. *Conservation Letters*, *14*(3), e12777. https://doi.org/10.1111/conl.12777

Larson, A. M., & Springer, J. (2016). *Recognition and Respect for Tenure Rights* (p. 20) [NRGF Conceptual Paper]. IUCN, CEESP and CIFOR.

Leberger, R., Rosa, I. M. D., Guerra, C. A., Wolf, F., & Pereira, H. M. (2020). Global patterns of forest loss across IUCN categories of protected areas. *Biological Conservation*, *241*, 108299. https://doi.org/10.1016/j.biocon.2019.108299

Leclère, D., Obersteiner, M., Barrett, M., Butchart, S. H. M., Chaudhary, A., De Palma, A., DeClerck, F. A. J., Di Marco, M., Doelman, J. C., Dürauer, M., Freeman, R., Harfoot, M., Hasegawa, T., Hellweg, S., Hilbers, J. P., Hill, S. L. L., Humpenöder, F., Jennings, N., Krisztin, T., … Young, L. (2020). Bending the curve of terrestrial biodiversity needs an integrated strategy. *Nature*, *585*(7826), 551–556. https://doi.org/10.1038/s41586-020-2705-y

Lee, T. M., Sigouin, A., Pinedo-Vasquez, M., & Nasi, R. (2020). The Harvest of Tropical Wildlife for Bushmeat and Traditional Medicine. *Annual Review of Environment and Resources*, *45*(1), 145–170. https://doi.org/10.1146/annurev-environ-102016-060827

Lemos, M. C., & Agrawal, A. (2006). Environmental Governance. *Annual Review of Environment and Resources*, *31*(1), 297–325. https://doi.org/10.1146/annurev.energy.31.042605.135621

Levers, C., Romero-Muñoz, A., Baumann, M., De Marzo, T., Fernández, P. D., Gasparri, N. I., Gavier-Pizarro, G. I., Waroux, Y. le P. de, Piquer-Rodríguez, M., Semper-Pascual, A., & Kuemmerle, T. (2021). Agricultural expansion and the ecological marginalization of forest-dependent people. *Proceedings of the National Academy of Sciences*, *118*(44), e2100436118. https://doi.org/10.1073/pnas.2100436118

Levis, C., Costa, F. R. C., Bongers, F., Peña-Claros, M., Clement, C. R., Junqueira, A. B., Neves, E. G., Tamanaha, E. K., Figueiredo, F. O. G., Salomão, R. P., Castilho, C. V., Magnusson, W. E., Phillips, O. L., Guevara, J. E., Sabatier, D., Molino, J.-F., López, D. C., Mendoza, A. M., Pitman, N. C. A., … ter Steege, H. (2017). Persistent effects of pre-Columbian plant domestication on Amazonian forest composition. *Science*, *355*(6328), 925–931. https://doi.org/10.1126/science.aal0157

Lloyd, C. T., Sorichetta, A., & Tatem, A. J. (2017). High resolution global gridded data for use in population studies. *Scientific Data*, *4*(1), 170001. https://doi.org/10.1038/sdata.2017.1

Loughlin, N. J. D., Gosling, W. D., Mothes, P., & Montoya, E. (2018). Ecological consequences of post-Columbian indigenous depopulation in the Andean–Amazonian corridor. *Nature Ecology & Evolution*, *2*(8), 1233–1236. https://doi.org/10.1038/s41559-018-0602-7

Luckeneder, S., Giljum, S., Schaffartzik, A., Maus, V., & Tost, M. (2021). Surge in global metal mining threatens vulnerable ecosystems. *Global Environmental Change*, *69*, 102303. https://doi.org/10.1016/j.gloenvcha.2021.102303

Lunstrum, E. (2013). Articulated sovereignty: Extending Mozambican state power through the Great Limpopo Transfrontier Park1. *Political Geography*, *36*, 1–11. https://doi.org/10.1016/j.polgeo.2013.04.003

Maezumi, S. Y., Alves, D., Robinson, M., de Souza, J. G., Levis, C., Barnett, R. L., Almeida de Oliveira, E., Urrego, D., Schaan, D., & Iriarte, J. (2018). The legacy of 4,500 years of polyculture agroforestry in the eastern Amazon. *Nature Plants*, *4*(8), 540–547. https://doi.org/10.1038/s41477-018-0205-y

Malavasi, M. (2020). The map of biodiversity mapping. *Biological Conservation*, *252*, 108843. https://doi.org/10.1016/j.biocon.2020.108843

Malhi, Y. (2012). The productivity, metabolism and carbon cycle of tropical forest vegetation: Carbon cycle of tropical forests. *Journal of Ecology*, *100*(1), 65–75. https://doi.org/10.1111/j.1365-2745.2011.01916.x

Malhi, Y. (2017). The Concept of the Anthropocene. *Annual Review of Environment and Resources*, *42*(1), 77–104. https://doi.org/10.1146/annurev-environ-102016-060854

Malhi, Y., Gardner, T. A., Goldsmith, G. R., Silman, M. R., & Zelazowski, P. (2014). Tropical Forests in the Anthropocene. *Annual Review of Environment and Resources*, *39*(1), 125–159. https://doi.org/10.1146/annurev-environ-030713-155141

Marques, A., Martins, I. S., Kastner, T., Plutzar, C., Theurl, M. C., Eisenmenger, N., Huijbregts, M. A. J., Wood, R., Stadler, K., Bruckner, M., Canelas, J., Hilbers, J. P., Tukker, A., Erb, K., & Pereira, H. M. (2019). Increasing impacts of land use on biodiversity and carbon sequestration driven by population and economic growth. *Nature Ecology & Evolution*, *3*(4), 628–637. https://doi.org/10.1038/s41559-019-0824-3

Martin, A., McGuire, S., & Sullivan, S. (2013). Global environmental justice and biodiversity conservation. *The Geographical Journal*, *179*(2), 122–131. https://doi.org/10.1111/geoj.12018

Mascia, M. B., Pailler, S., Krithivasan, R., Roshchanka, V., Burns, D., Mlotha, M. J., Murray, D. R., & Peng, N. (2014). Protected area downgrading, downsizing, and degazettement (PADDD) in Africa, Asia, and Latin America and the Caribbean, 1900–2010. *Biological Conservation*, *169*, 355–361. https://doi.org/10.1016/j.biocon.2013.11.021

Maxwell, S. L., Cazalis, V., Dudley, N., Hoffmann, M., Rodrigues, A. S. L., Stolton, S., Visconti, P., Woodley, S., Kingston, N., Lewis, E., Maron, M., Strassburg, B. B. N., Wenger, A., Jonas, H. D., Venter, O., & Watson, J. E. M. (2020). Area-based conservation in the twenty-first century. *Nature*, *586*(7828), 217–227. https://doi.org/10.1038/s41586-020-2773-z

McDermott, C. L., Montana, J., Bennett, A., Gueiros, C., Hamilton, R., Hirons, M., Maguire-Rajpaul, V. A., Parry, E., & Picot, L. (2022). Transforming land use governance: Global targets without equity miss the mark. *Environmental Policy and Governance*, eet.2027. https://doi.org/10.1002/eet.2027

Meinherz, F., Fritz, L., & Schneider, F. (2020). 3—How values play into sustainability assessments: Challenges and a possible way forward. In C. R. Binder, R. Wyss, & E. Massaro (Eds.), *Sustainability Assessment of Urban Systems* (1st ed., pp. 65–86). Cambridge University Press. https://doi.org/10.1017/9781108574334

Merçon, J., Vetter, S., Tengö, M., Cocks, M., Balvanera, P., Rosell, J. A., & Ayala-Orozco, B. (2019). From local landscapes to international policy: Contributions of the biocultural paradigm to global sustainability. *Global Sustainability*, *2*, e7. https://doi.org/10.1017/sus.2019.4

Mi, C., Ma, L., & Yang, M. (2023). Global Protected Areas as refuges for amphibians and reptiles under climate change. *Nature Communications*, *14*, 1389. https://doi.org/10.1038/s41467-023-36987-y

Michon, G., de Foresta, H., Levang, P., & Verdeaux, F. (2007). Domestic Forests: A New Paradigm for Integrating Local Communities' Forestry into Tropical Forest Science. *Ecology and Society*, *12*(2), 1.

Miettinen, J., Stibig, H.-J., & Achard, F. (2014). Remote sensing of forest degradation in Southeast Asia—Aiming for a regional view through 5–30 m satellite data. *Global Ecology and Conservation*, *2*, 24–36. https://doi.org/10.1016/j.gecco.2014.07.007

Millenium Ecosystem Assessment. (2005). *Ecosystems and Human Well-being: Biodiversity synthesis*. World Resources Institute. https://www.millenniumassessment.org/documents/document.356.aspx.pdf

Milodowski, D. T., Coomes, D. A., Swinfield, T., Jucker, T., Riutta, T., Malhi, Y., Svátek, M., Kvasnica, J., Burslem, D. F. R. P., Ewers, R. M., Teh, Y. A., & Williams, M. (2021). The impact of logging on vertical canopy structure across a gradient of tropical forest degradation intensity in Borneo. *Journal of Applied Ecology*, *58*(8), 1764–1775. https://doi.org/10.1111/1365-2664.13895

Moon, K., & Blackman, D. (2014). A Guide to Understanding Social Science Research for Natural Scientists: Social Science for Natural Scientists. *Conservation Biology*, *28*(5), 1167–1177. https://doi.org/10.1111/cobi.12326

Moon, K., Blackman, D. A., Adams, V. M., Colvin, R. M., Davila, F., Evans, M. C., Januchowski-Hartley, S. R., Bennett, N. J., Dickinson, H., Sandbrook, C., Sherren, K., St. John, F. A. V., Van Kerkhoff, L., & Wyborn, C. (2019). Expanding the role of social science in conservation through an engagement with philosophy, methodology, and methods. *Methods in Ecology and Evolution*, *10*(3), 294–302. https://doi.org/10.1111/2041-210X.13126

Moore, J., W. (Ed.). (2016). *Anthropocene or Capitalocene? Nature, History, and the Crisis of Capitalism*. PM Press.

Moranta, J., Torres, C., Murray, I., Hidalgo, M., Hinz, H., & Gouraguine, A. (2021). Transcending capitalism growth strategies for biodiversity conservation. *Conservation Biology*, cobi.13821. https://doi.org/10.1111/cobi.13821

Narain, D., Sonter, L. J., Lechner, A. M., Watson, J. E. M., Simmonds, J. S., & Maron, M. (2023). Global assessment of the biodiversity safeguards of development banks that finance infrastructure. *Conservation Biology*, e14095. https://doi.org/10.1111/cobi.14095

Naughton-Treves, L., Holland, M. B., & Brandon, K. (2005). The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Environment and Resources*, *30*(1), 219–252. https://doi.org/10.1146/annurev.energy.30.050504.164507

Nelson, A., & Chomitz, K. M. (2011). Effectiveness of Strict vs. Multiple Use Protected Areas in Reducing Tropical Forest Fires: A Global Analysis Using Matching Methods. *PLoS ONE*, *6*(8), e22722. https://doi.org/10.1371/journal.pone.0022722

Nelson, A., Weiss, D. J., van Etten, J., Cattaneo, A., McMenomy, T. S., & Koo, J. (2019). A suite of global accessibility indicators. *Scientific Data*, *6*(1), 266. https://doi.org/10.1038/s41597-019-0265-5

Nepstad, D., Schwartzman, S., Bamberger, B., Santilli, M., Ray, D., Schlesinger, P., Lefebvre, P., Alencar, A., Prinz, E., Fiske, G., & Rolla, A. (2006). Inhibition of Amazon Deforestation and Fire by Parks and Indigenous Lands: Inhibition of Amazon Deforestation and Fire. *Conservation Biology*, *20*(1), 65–73. https://doi.org/10.1111/j.1523-1739.2006.00351.x

Nepstad, S., & Bob, C. (2006). When Do Leaders Matter? Hypotheses on Leadership Dynamics in Social Movements. *Mobilization: An International Quarterly*, *11*(1), 1–22. https://doi.org/10.17813/maiq.11.1.013313600164m727

Neumann, R. P. (2003). The Production of Nature: Colonial Recasting of African Landscape in Serengeti National Park. In K. S. Zimmerer & T. J. Bassett (Eds.), *Political Ecology: An Integrative Approach to Geography and Environment-Development Studies* (p. 310). Guildford Press.

Newbold, T., Hudson, L. N., Arnell, A. P., Contu, S., De Palma, A., Ferrier, S., Hill, S. L. L., Hoskins, A. J., Lysenko, I., Phillips, H. R. P., Burton, V. J., Chng, C. W. T., Emerson, S., Gao, D., Pask-Hale, G., Hutton, J., Jung, M., Sanchez-Ortiz, K., Simmons, B. I., … Purvis, A. (2016). Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. *Science*, *353*(6296), 288–291. https://doi.org/10.1126/science.aaf2201

Nolte, C., Agrawal, A., Silvius, K. M., & Soares-Filho, B. S. (2013). Governance regime and location influence avoided deforestation success of protected areas in the Brazilian Amazon. *Proceedings of the National Academy of Sciences*, *110*(13), 4956–4961. https://doi.org/10.1073/pnas.1214786110

Noss, A. J., & Leny Cuellar, R. (2008). The sustainability of hunting Tapirus terrestris and Tayassu pecari in the Isoso indigenous communal land: The unified harvest model. *Mastozoología Neotropical*, *15*(2), 241–252.

O’Bryan, C. J., Garnett, S. T., Fa, J. E., Leiper, I., Rehbein, J., Fernández-Llamazares, Á., Jackson, M. V., Jonas, H. D., Brondizio, E. S., Burgess, N. D., Robinson, C. J., Zander, K. K., Venter, O., & Watson, J. E. M. (2020). The importance of indigenous peoples’ lands for the conservation of terrestrial mammals. *Conservation Biology*, cobi.13620. https://doi.org/10.1111/cobi.13620

Obura, D. O., Katerere, Y., Mayet, M., Kaelo, D., Msweli, S., Mather, K., Harris, J., Louis, M., Kramer, R., Teferi, T., Samoilys, M., Lewis, L., Bennie, A., Kumah, F., Isaacs, M., & Nantongo, P. (2021). Integrate biodiversity targets from local to global levels. *Science*, *373*(6556), 746–748. https://doi.org/10.1126/science.abh2234

OHCHR Press Release. (2021). *Thailand: UN experts warn against heritage status for Kaeng Krachan national park*. https://www.ohchr.org/en/press-releases/2021/07/thailand-un-experts-warn-against-heritage-status-kaeng-krachan-national-park

Oliveira, P. J. C., Asner, G. P., Knapp, D. E., Almeyda, A., Galván-Gildemeister, R., Keene, S., Raybin, R. F., & Smith, R. C. (2007). Land-Use Allocation Protects the Peruvian Amazon. *Science*, *317*(5842), 1233–1236. https://doi.org/10.1126/science.1146324

Owen, J. R., Kemp, D., Lechner, A. M., Harris, J., Zhang, R., & Lèbre, É. (2022). Energy transition minerals and their intersection with land-connected peoples. *Nature Sustainability*, *6*(2), 203–211. https://doi.org/10.1038/s41893-022-00994-6

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., Phillips, O. L., Shvidenko, A., Lewis, S. L., Canadell, J. G., Ciais, P., Jackson, R. B., Pacala, S. W., McGuire, A. D., Piao, S., Rautiainen, A., Sitch, S., & Hayes, D. (2011). A Large and Persistent Carbon Sink in the World's Forests. *Science*, *333*(6045), 988–993. https://doi.org/10.1126/science.1201609

Paulson, N., Laudati, A., Doolittle, A., Welch-Devine, M., & Pena, P. (2012). Indigenous Peoples' Participation in Global Conservation: Looking beyond Headdresses and Face Paint. *Environmental Values*, *21*(3), 255–276. https://doi.org/10.3197/096327112X13400390125894

Pearson, T. R. H., Brown, S., Murray, L., & Sidman, G. (2017). Greenhouse gas emissions from tropical forest degradation: An underestimated source. *Carbon Balance and Management*, *12*(1), 3. https://doi.org/10.1186/s13021-017-0072-2

Pebesma, E. (2018). Simple Features for R: Standardized Support for Spatial Vector Data. *The R Journal*, *10*(1), 439–446. https://doi.org/10.32614/RJ-2018-009

Peluso, N. L. (1993). Coercing conservation? *Global Environmental Change*, *3*(2), 199–217. https://doi.org/10.1016/0959-3780(93)90006-7

Peluso, N. L., & Vandergeest, P. (2001). Genealogies of the Political Forest and Customary Rights in Indonesia, Malaysia, and Thailand. *The Journal of Asian Studies*, *60*(3), 761–812. https://doi.org/10.2307/2700109

Pendrill, F., Gardner, T. A., Meyfroidt, P., Persson, U. M., Adams, J., Azevedo, T., Bastos Lima, M. G., Baumann, M., Curtis, P. G., De Sy, V., Garrett, R., Godar, J., Goldman, E. D., Hansen, M. C., Heilmayr, R., Herold, M., Kuemmerle, T., Lathuillière, M. J., Ribeiro, V., … West, C. (2022). Disentangling the numbers behind agriculture-driven tropical deforestation. *Science*, *377*(6611), eabm9267. https://doi.org/10.1126/science.abm9267

Pfaff, A., Robalino, J., Lima, E., Sandoval, C., & Herrera, L. D. (2014). Governance, Location and Avoided Deforestation from Protected Areas: Greater Restrictions Can Have Lower Impact, Due to Differences in Location. *World Development*, *55*, 7–20. https://doi.org/10.1016/j.worlddev.2013.01.011

Pimm, S. L., & Raven, P. (2000). Extinction by numbers. *Nature*, *403*(6772), 843–845. https://doi.org/10.1038/35002708

Pin, C., Kamler, J. F., Toem, Y., Lay, D., Vorn, K., Kim, N., & Macdonald, D. W. (2022). First record of a giant muntjac *Muntiacus vuquangensis* (Cervidae) from Cambodia. *Mammalia*, *86*(5), 434–438. https://doi.org/10.1515/mammalia-2021-0132

Plumptre, A. J., Baisero, D., Belote, R. T., Vázquez-Domínguez, E., Faurby, S., Jędrzejewski, W., Kiara, H., Kühl, H., Benítez-López, A., Luna-Aranguré, C., Voigt, M., Wich, S., Wint, W., Gallego-Zamorano, J., & Boyd, C. (2021). Where Might We Find Ecologically Intact Communities? *Frontiers in Forests and Global Change*, *4*, 626635. https://doi.org/10.3389/ffgc.2021.626635

Pörtner, H.-O., Scholes, R. J., Arneth, A., Barnes, D. K. A., Burrows, M. T., Diamond, S. E., Duarte, C. M., Kiessling, W., Leadley, P., Managi, S., McElwee, P., Midgley, G., Ngo, H. T., Obura, D., Pascual, U., Sankaran, M., Shin, Y. J., & Val, A. L. (2023). Overcoming the coupled climate and biodiversity crises and their societal impacts. *Science*, *380*(6642), eabl4881. https://doi.org/10.1126/science.abl4881

Potapov, P., Hansen, M. C., Laestadius, L., Turubanova, S., Yaroshenko, A., Thies, C., Smith, W., Zhuravleva, I., Komarova, A., Minnemeyer, S., & Esipova, E. (2017). The last frontiers of wilderness: Tracking loss of intact forest landscapes from 2000 to 2013. *Science Advances*, *3*(1), e1600821. https://doi.org/10.1126/sciadv.1600821

Pyhälä, A., Orozco, A. O., & Counsell, S. (2016). *Protected Areas in the Congo Basin: Failing both people and biodiversity?* (Under the Canopy). Rainforest Foundation UK.

QGIS. (2020). *QGIS Geographic Information System* (3.4.6) [Computer software]. Open Source Geospatial Foundation Project. http://qgis.org

Quijano Vallejos, P., Veit, P., Tipula, P., & Reytar, K. (2020). Undermining Rights: Indigenous Lands and Mining in the Amazon. *World Resources Institute*. https://doi.org/10.46830/wrirpt.19.00085

R Core Team. (2021). *R: A language and environment for statistical computing.* (4.1.1) [Computer software]. R Foundation for Statistical Computing. https://www.R-project.org/

Rainforest Foundation UK, Minority Rights Group International, Survival International, Coorg Organisation for Rural Development, KHOJ Melghat, Katiba Institute, Ogiek Peoples' Development Program, ProDESC, Comisión Mexicana de Defensa y Promoción de los Derechos Humanos, & Dynamique Mondiale des Jeunes. (2021, March 8). *NGO concerns over the proposed 30% target for protected areas and absence of safeguards for Indigenous Peoples and local communities*. https://minorityrights.org/wp-content/uploads/2021/04/en-fr-es-it-de-200928.pdf

Ramirez Cover, A. (2017). *A political ecology of neoliberal multiculturalism social inclusion and market-based conservation in indigenous Costa Rica*. Erasmus University Rotterdam.

Redford, K. H. (1992). The Empty Forest. *BioScience*, *42*(6), 412–422. https://doi.org/10.2307/1311860

Redford, K. H., Coppolillo, P., Sanderson, E. W., Da Fonseca, G. A. B., Dinerstein, E., Groves, C., Mace, G., Maginnis, S., Mittermeier, R. A., Noss, R., Olson, D., Robinson, J. G., Vedder, A., & Wright, M. (2003). Mapping the Conservation Landscape. *Conservation Biology*, *17*(1), 116–131. https://doi.org/10.1046/j.1523-1739.2003.01467.x

Renwick, A. R., Robinson, C. J., Garnett, S. T., Leiper, I., Possingham, H. P., & Carwardine, J. (2017). Mapping Indigenous land management for threatened species conservation: An Australian case-study. *PLOS ONE*, *12*(3), e0173876. https://doi.org/10.1371/journal.pone.0173876

Resende, F. M., Meyer, L., Vieira, R. R. S., Bizri, H. R. E., Pacheco, A. A., Fernandes, G. W., & Loyola, R. (2021). *Covid-19 pandemic and harmful policies push Brazil into an environmental crisis* [Preprint]. Open Science Framework. https://doi.org/10.31219/osf.io/w4jpn

Reyes-García, V., Fernández-Llamazares, Á., Aumeeruddy-Thomas, Y., Benyei, P., Bussmann, R. W., Diamond, S. K., García-del-Amo, D., Guadilla-Sáez, S., Hanazaki, N., Kosoy, N., Lavides, M., Luz, A. C., McElwee, P., Meretsky, V. J., Newberry, T., Molnár, Z., Ruiz-Mallén, I., Salpeteur, M., Wyndham, F. S., … Brondizio, E. S. (2021). Recognizing Indigenous peoples' and local communities' rights and agency in the post-2020 Biodiversity Agenda. *Ambio*. https://doi.org/10.1007/s13280-021-01561-7

Riggio, J., Baillie, J. E. M., Brumby, S., Ellis, E., Kennedy, C. M., Oakleaf, J. R., Tait, A., Tepe, T., Theobald, D. M., Venter, O., Watson, J. E. M., & Jacobson, A. P. (2020). Global human influence maps reveal clear opportunities in conserving Earth's remaining intact terrestrial ecosystems. *Global Change Biology*, *26*(8), 4344–4356. https://doi.org/10.1111/gcb.15109

Rights and Resources Initiative. (2015). *Who owns the world's land? A global baseline of formally recognized indigenous and community land rights*. RRI. https://rightsandresources.org/wp-content/uploads/GlobalBaseline_web.pdf

Rights and Resources Initiative. (2020). *Estimate of the area of land and territories of Indigenous Peoples, local communities, and Afro-descendants where their rights have not been recognized* (p. 32) [Technical Report]. Rights and Resources Initiative. https://rightsandresources.org/publication/estimate-of-the-area-of-land-and-territories-of-indigenous-peoples-local-communities-and-afro-descendants-where-their-rights-have-not-been-recognized

Robbins, P. (2007). Fortress Conservation. In *Encyclopedia of Environment and Society*. SAGE Publications, Inc.

Roberts, P., Hunt, C., Arroyo-Kalin, M., Evans, D., & Boivin, N. (2017). The deep human prehistory of global tropical forests and its relevance for modern conservation. *Nature Plants*, *3*(8), 17093. https://doi.org/10.1038/nplants.2017.93

Robinson, B. E., Holland, M. B., & Naughton-Treves, L. (2014). Does secure land tenure save forests? A meta-analysis of the relationship between land tenure and tropical deforestation. *Global Environmental Change*, *29*, 281–293. https://doi.org/10.1016/j.gloenvcha.2013.05.012

Robinson, B. E., Masuda, Y. J., Kelly, A., Holland, M. B., Bedford, C., Childress, M., Fletschner, D., Game, E. T., Ginsburg, C., Hilhorst, T., Lawry, S., Miteva, D. A., Musengezi, J., Naughton-Treves, L., Nolte, C., Sunderlin, W. D., & Veit, P. (2018). Incorporating Land Tenure Security into Conservation: Conservation and land tenure security. *Conservation Letters*, *11*(2), e12383. https://doi.org/10.1111/conl.12383

Ross, N. (2020). *Fasterize: Fast Polygon to Raster Conversion. R package version 1.0.0* [Computer software]. https://CRAN.R-project.org/package=fasterize

Rubis, J. M. (2020). The orang utan is not an indigenous name: Knowing and naming the maias as a decolonizing epistemology. *Cultural Studies*, *34*(5), 811–830. https://doi.org/10.1080/09502386.2020.1780281

Rubis, J. M., & Theriault, N. (2020). Concealing protocols: Conservation, Indigenous survivance, and the dilemmas of visibility. *Social & Cultural Geography*, *21*(7), 962–984. https://doi.org/10.1080/14649365.2019.1574882

Rudel, T. K. (2009). How Do People Transform Landscapes? A Sociological Perspective on Suburban Sprawl and Tropical Deforestation. *American Journal of Sociology*, *115*(1), 129–154. https://doi.org/10.1086/597794

Ruggiero, P. G. C., Pfaff, A., Nichols, E., Rosa, M., & Metzger, J. P. (2021). Election cycles affect deforestation within Brazil's Atlantic Forest. *Conservation Letters*, *14*(5). https://doi.org/10.1111/conl.12818

Saif, O., Keane, A., & Staddon, S. (2022). Making a case for the consideration of trust, justice, and power in conservation relationships. *Conservation Biology*, *36*(4). https://doi.org/10.1111/cobi.13903

Salmón, E. (2000). Kincentric Ecology: Indigenous perceptions of the human-nature relationship. *Ecological Applications*, *10*(5), 1327–1332. https://doi.org/10.1890/1051-0761(2000)010[1327:KEIPOT]2.0.CO;2

Sandbrook, C., Albury-Smith, S., Allan, J. R., Bhola, N., Bingham, H. C., Brockington, D., Byaruhanga, A. B., Fajardo, J., Fitzsimons, J., Franks, P., Fleischman, F., Frechette, A., Kakuyo, K., Kaptoyo, E., Kuemmerle, T., Kalunda, P. N., Nuvunga, M., O’Donnell, B., Onyai, F., … Zaehringer, J. G. (2023). Social considerations are crucial to success in implementing the 30×30 global conservation target. *Nature Ecology & Evolution*. https://doi.org/10.1038/s41559-023-02048-2

Sasaoka, M., & Laumonier, Y. (2012). Suitability of Local Resource Management Practices Based on Supernatural Enforcement Mechanisms in the Local Social-cultural Context. *Ecology and Society*, *17*(4), 6. https://doi.org/10.5751/ES-05124-170406

Sayre, N. F., Davis, D. K., Bestelmeyer, B., & Williamson, J. C. (2017). Rangelands: Where Anthromes Meet Their Limits. *Land*, *6*(2), 31. https://doi.org/10.3390/land6020031

Scheidel, A., Del Bene, D., Liu, J., Navas, G., Mingorría, S., Demaria, F., Avila, S., Roy, B., Ertör, I., Temper, L., & Martínez-Alier, J. (2020). Environmental conflicts and defenders: A global overview. *Global Environmental Change*, *63*, 102104. https://doi.org/10.1016/j.gloenvcha.2020.102104

Scheidel, A., Fernández-Llamazares, Á., Bara, A. H., Del Bene, D., David-Chavez, D. M., Fanari, E., Garba, I., Hanaček, K., Liu, J., Martínez-Alier, J., Navas, G., Reyes-García, V., Roy, B., Temper, L., Thiri, M. A., Tran, D., Walter, M., & Whyte, K. P. (2023). Global impacts of extractive and industrial development projects on Indigenous Peoples’ lifeways, lands, and rights. *Science Advances*, *9*(23), eade9557. https://doi.org/10.1126/sciadv.ade9557

Schleicher, J., Eklund, J., D. Barnes, M., Geldmann, J., Oldekop, J. A., & Jones, J. P. G. (2020). Statistical matching for conservation science. *Conservation Biology*, *34*(3), 538–549. https://doi.org/10.1111/cobi.13448

Schleicher, J., Peres, C. A., Amano, T., Llactayo, W., & Leader-Williams, N. (2017). Conservation performance of different conservation governance regimes in the Peruvian Amazon. *Scientific Reports*, *7*(1), 11318. https://doi.org/10.1038/s41598-017-10736-w

Schuster, R., Germain, R. R., Bennett, J. R., Reo, N. J., & Arcese, P. (2019). Vertebrate biodiversity on indigenous-managed lands in Australia, Brazil, and Canada equals that in protected areas. *Environmental Science & Policy*, *101*, 1–6. https://doi.org/10.1016/j.envsci.2019.07.002

Schwartzman, S., Boas, A. V., Ono, K. Y., Fonseca, M. G., Doblas, J., Zimmerman, B., Junqueira, P., Jerozolimski, A., Salazar, M., Junqueira, R. P., & Torres, M. (2013). The natural and social history of the indigenous lands and protected areas corridor of the Xingu River basin. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *368*(1619), 20120164. https://doi.org/10.1098/rstb.2012.0164

Segan, D. B., Murray, K. A., & Watson, J. E. M. (2016). A global assessment of current and future biodiversity vulnerability to habitat loss–climate change interactions. *Global Ecology and Conservation*, *5*, 12–21. https://doi.org/10.1016/j.gecco.2015.11.002

Selibas, D. (2022, April 11). Ecuador's Pastaza province, Indigenous groups collaborate on forest conservation. *Mongabay*. https://news.mongabay.com/2022/04/ecuadors-pastaza-province-indigenous-groups-collaborate-on-forest-conservation/

Senior, R. A., Hill, J. K., & Edwards, D. P. (2019). Global loss of climate connectivity in tropical forests. *Nature Climate Change*, *9*(8), 623–626. https://doi.org/10.1038/s41558-019-0529-2

Seymour, F., La Vina, T., & Hite, K. (2014). *Evidence linking community-level tenure and forest condition: An annotated bibliography* (p. 61). Climate and Land Use Alliance.

Shackleton, R. T., Walters, G., Bluwstein, J., Djoudi, H., Fritz, L., Lafaye De Micheaux, F., Loloum, T., Nguyen, V. T. H., Rann Andriamahefazafy, M., Sithole, S. S., & Kull, C. A. (2023). Navigating power in conservation. *Conservation Science and Practice*, *5*(3). https://doi.org/10.1111/csp2.12877

Sheil, D., Boissière, M., & Beaudoin, G. (2015). Unseen sentinels: Local monitoring and control in conservation's blind spots. *Ecology and Society*, *20*(2), 39. https://doi.org/10.5751/ES-07625-200239

Simkin, R. D., Seto, K. C., McDonald, R. I., & Jetz, W. (2022). Biodiversity impacts and conservation implications of urban land expansion projected to 2050. *Proceedings of the National Academy of Sciences*, *119*(12), e2117297119. https://doi.org/10.1073/pnas.2117297119

Smith, D. A., Holland, M. B., Michon, A., Ibáñez, A., & Herrera, F. (2017). The hidden layer of indigenous land tenure: Informal forest ownership and its implications for forest use and conservation in Panama's largest collective territory. *International Forestry Review*, *19*(4), 478–494.

Soto-Navarro, C., Ravilious, C., Arnell, A., De Lamo, X., Harfoot, M., Hill, S. L. L., Wearn, O. R., Santoro, M., Bouvet, A., Mermoz, S., Le Toan, T., Xia, J., Liu, S., Yuan, W., Spawn, S. A., Gibbs, H. K., Ferrier, S., Harwood, T., Alkemade, R., … Kapos, V. (2020). Mapping co-benefits for carbon storage and biodiversity to inform conservation policy and action. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *375*(1794), 20190128. https://doi.org/10.1098/rstb.2019.0128

Spooner, F. E. B., Pearson, R. G., & Freeman, R. (2018). Rapid warming is associated with population decline among terrestrial birds and mammals globally. *Global Change Biology*, *24*(10), 4521–4531. https://doi.org/10.1111/gcb.14361

Spracklen, B. D., Kalamandeen, M., Galbraith, D., Gloor, E., & Spracklen, D. V. (2015). A Global Analysis of Deforestation in Moist Tropical Forest Protected Areas. *PLOS ONE*, *10*(12), e0143886. https://doi.org/10.1371/journal.pone.0143886

Stevens, S., Broome, Neema Pathak, & Jaegar, T. (2016). *Recognising and Respecting ICCAs Overlapped by Protected Areas* (Report for the ICCA Consortium). https://www.iccaconsortium.org/wp-content/uploads/2016/11/publication-Recognising-and-Respecting-ICCAs-Overlapped-by-PAs-Stevens-et-al-2016-en.pdf

Stolton, S., & Dudley, N. (1999). A preliminary survey of management effectiveness status and threats in forest protected areas. *Parks*, *9*(2), 27–33.

Stuart, E. A., & Rubin, D. B. (2008). Best Practices in Quasi–Experimental Designs: Matching Methods for Causal Inference. In J. Osborne, *Best Practices in Quantitative Methods* (pp. 155–176). SAGE Publications, Inc. https://doi.org/10.4135/9781412995627.d14

Sunkar, A., & Santosa, Y. (2018). Biodiversity conservation at Betung Kerihun National Park: The positive role of customary law in mitigating a local community trust deficit. *IOP Conference Series: Earth and Environmental Science*, *196*, 012024. https://doi.org/10.1088/1755-1315/196/1/012024

Sze, J. S., Carrasco, L. R., Childs, D., & Edwards, D. P. (2022). Reduced deforestation and degradation in Indigenous Lands pan-tropically. *Nature Sustainability*, *5*(2), 123–130. https://doi.org/10.1038/s41893-021-00815-2

Sze, J. S., Childs, D. Z., Carrasco, L. R., & Edwards, D. P. (2022). Indigenous lands in protected areas have high forest integrity across the tropics. *Current Biology*, *32*(22), 4949-4956.e3. https://doi.org/10.1016/j.cub.2022.09.040

Tauli-Corpuz, V., Alcorn, J., Molnar, A., Healy, C., & Barrow, E. (2020). Cornered by PAs: Adopting rights-based approaches to enable cost-effective conservation and climate action. *World Development*, *130*, 104923. https://doi.org/10.1016/j.worlddev.2020.104923

Tellman, B., Sesnie, S. E., Magliocca, N. R., Nielsen, E. A., Devine, J. A., McSweeney, K., Jain, M., Wrathall, D. J., Dávila, A., Benessaiah, K., & Aguilar-Gonzalez, B. (2020). Illicit

Drivers of Land Use Change: Narcotrafficking and Forest Loss in Central America. *Global Environmental Change*, *63*, 102092. https://doi.org/10.1016/j.gloenvcha.2020.102092

Tracewski, Ł., Butchart, S. H. M., Di Marco, M., Ficetola, G. F., Rondinini, C., Symes, A., Wheatley, H., Beresford, A. E., & Buchanan, G. M. (2016). Toward quantification of the impact of 21st-century deforestation on the extinction risk of terrestrial vertebrates: Effects of Deforestation on Vertebrates. *Conservation Biology*, *30*(5), 1070–1079. https://doi.org/10.1111/cobi.12715

Tran, T. C., Ban, N. C., & Bhattacharyya, J. (2020). A review of successes, challenges, and lessons from Indigenous protected and conserved areas. *Biological Conservation*, *241*, 108271. https://doi.org/10.1016/j.biocon.2019.108271

Transparent World, & Global Forest Watch. (2019). *Tree Plantations* [Computer software].

Tregidgo, D. J., Barlow, J., Pompeu, P. S., De Almeida Rocha, M., & Parry, L. (2017). Rainforest metropolis casts 1,000-km defaunation shadow. *Proceedings of the National Academy of Sciences*, *114*(32), 8655–8659. https://doi.org/10.1073/pnas.1614499114

Tseng, T.-W. J., Robinson, B. E., Bellemare, M. F., BenYishay, A., Blackman, A., Boucher, T., Childress, M., Holland, M. B., Kroeger, T., Linkow, B., Diop, M., Naughton, L., Rudel, T., Sanjak, J., Shyamsundar, P., Veit, P., Sunderlin, W., Zhang, W., & Masuda, Y. J. (2021). Influence of land tenure interventions on human well-being and environmental outcomes. *Nature Sustainability*, *4*, 242–251. https://doi.org/10.1038/s41893-020-00648-5

Tuhiwai Smith, L. (2012). *Decolonizing Methodologies: Research and Indigenous Peoples* (2nd ed.). Zen Books Ltd.

Tulone, A., Galati, A., Pecoraro, S., Carroccio, A., Siggia, D., Virzì, M., & Crescimanno, M. (2022). Main intrinsic factors driving land grabbing in the African countries' agro-food industry. *Land Use Policy*, *120*, 106225. https://doi.org/10.1016/j.landusepol.2022.106225

UN News. (2022). *Tanzania: More violence feared over bid to evict Maasai from ancestral lands*. https://news.un.org/en/story/2022/06/1120442

UNEP-WCMC, & IUCN. (2020). *Protected Planet: The World Database on Protected Areas (WDPA) Online, January 2020*. UNEP-WCMC and IUCN. www.protectedplanet.net

UNEP-WCMC, & IUCN. (2023). *Protected Planet: The World Database on Protected Areas (WDPA) and World Database on Other Effective Area-based Conservation Measures (WD-OECM) [Online], May 2023*. UNEP-WCMC and IUCN. http://protectedplanet.net/

United Nations. (2007). *United Nations Declaration of the Rights of Indigenous Peoples*. https://www.un.org/development/desa/indigenouspeoples/wp-content/uploads/sites/19/2018/11/UNDRIP_E_web.pdf

United Nations. (2009). *State of the World's Indigenous Peoples* (ST/ESA/328). Secretariat of the Permanent Forum on Indigenous Issues, Division for Social Policy and Development, Department of Economic and Social Affairs, United Nations.

United Nations, Department of Economic and Social Affairs, United Nations, Division for Inclusive Social Development, Permanent Forum on Indigenous Issues (United Nations), & Secretariat. (2021). *State of the world's indigenous peoples: Rights to lands, territories and resources*.

Vancutsem, C., Achard, F., Pekel, J.-F., Vieilledent, G., Carboni, S., Simonetti, D., Gallego, J., Aragão, L. E. O. C., & Nasi, R. (2021). Long-term (1990–2019) monitoring of forest cover changes in the humid tropics. *Science Advances*, *7*(10), eabe1603. https://doi.org/10.1126/sciadv.abe1603

Veber, H., Dahl, J., Wilson, F., & Wæhle, E. (Eds.). (1993). *'... Never drink from the same cup': Proceedings of the conference on Indigenous Peoples in Africa* (73; p. 329). IWGIA. https://www.iwgia.org/images/publications/0153_74_Never_drink.pdf

Vergara-Asenjo, G., & Potvin, C. (2014). Forest protection and tenure status: The key role of indigenous peoples and protected areas in Panama. *Global Environmental Change*, *28*, 205–215. https://doi.org/10.1016/j.gloenvcha.2014.07.002

Walker, R., & Simmons, C. (2018). Endangered Amazon: An Indigenous Tribe Fights Back Against Hydropower Development in the Tapajós Valley. *Environment: Science and Policy for Sustainable Development*, *60*(2), 4–15. https://doi.org/10.1080/00139157.2018.1418994

Walker, W. S., Gorelik, S. R., Baccini, A., Aragon-Osejo, J. L., Josse, C., Meyer, C., Macedo, M. N., Augusto, C., Rios, S., Katan, T., de Souza, A. A., Cuellar, S., Llanos, A., Zager, I., Mirabal, G. D., Solvik, K. K., Farina, M. K., Moutinho, P., & Schwartzman, S. (2020). The role of forest conversion, degradation, and disturbance in the carbon dynamics of Amazon indigenous territories and protected areas. *Proceedings of the National Academy of Sciences*, *117*(6), 3015–3025. https://doi.org/10.1073/pnas.1913321117

Wang, M. M. H., Carrasco, L. R., & Edwards, D. P. (2020). Reconciling Rubber Expansion with Biodiversity Conservation. *Current Biology*, *30*(19), R1073–R1075. https://doi.org/10.1016/j.cub.2020.07.014

Warren-Thomas, E., Dolman, P. M., & Edwards, D. P. (2015). Increasing Demand for Natural Rubber Necessitates a Robust Sustainability Initiative to Mitigate Impacts on Tropical Biodiversity: Rubber sustainability and biodiversity. *Conservation Letters*, *8*(4), 230–241. https://doi.org/10.1111/conl.12170

Watson, J. E. M., Dudley, N., Segan, D. B., & Hockings, M. (2014). The performance and potential of protected areas. *Nature*, *515*(7525), 67–73. https://doi.org/10.1038/nature13947

Watson, J. E. M., Evans, T., Venter, O., Williams, B., Tulloch, A., Stewart, C., Thompson, I., Ray, J. C., Murray, K., Salazar, A., McAlpine, C., Potapov, P., Walston, J., Robinson, J. G., Painter, M., Wilkie, D., Filardi, C., Laurance, W. F., Houghton, R. A., … Lindenmayer, D. (2018). The exceptional value of intact forest ecosystems. *Nature Ecology & Evolution*, *2*(4), 599–610. https://doi.org/10.1038/s41559-018-0490-x

Weinzettel, J., Hertwich, E. G., Peters, G. P., Steen-Olsen, K., & Galli, A. (2013). Affluence drives the global displacement of land use. *Global Environmental Change*, *23*(2), 433–438. https://doi.org/10.1016/j.gloenvcha.2012.12.010

Weisse, M., & Goldman, E. D. (2020, June 2). We Lost a Football Pitch of Primary Rainforest Every 6 Seconds in 2019. *World Resources Institute*. https://www.wri.org/blog/2020/06/global-tree-cover-loss-data-2019

Wiedmann, T., Lenzen, M., Keyßer, L. T., & Steinberger, J. K. (2020). Scientists' warning on affluence. *Nature Communications*, *11*(1), 3107. https://doi.org/10.1038/s41467-020-16941-y

Wilcove, D. S., & Koh, L. P. (2010). Addressing the threats to biodiversity from oil-palm agriculture. *Biodiversity and Conservation*, *19*(4), 999–1007. https://doi.org/10.1007/s10531-009-9760-x

Witter, R., & Satterfield, T. (2019). The Ebb and Flow of Indigenous Rights Recognitions in Conservation Policy: Indigenous Rights Recognitions in Conservation Policy. *Development and Change*, *50*(4), 1083–1108. https://doi.org/10.1111/dech.12456

Wolf, C., Levi, T., Ripple, W. J., Zárrate-Charry, D. A., & Betts, M. G. (2021). A forest loss report card for the world's protected areas. *Nature Ecology & Evolution*, *5*(4), 520–529. https://doi.org/10.1038/s41559-021-01389-0

Wood, S. (2019). *mgcv: Mixed GAM computation vehicle with automatic smoothness estimation* [Manual]. https://CRAN.R-project.org/package=mgcv

WWF. (2022). *Living Planet Report 2022—Building a nature positive society*. WWF International. https://www.wwf.org.uk/sites/default/files/2023-05/WWF-Living-Planet-Report-2022.pdf

WWF, UNEP-WCMC, SGP/ICCA-GSI, LM, TNC, CI, WCS, EP, ILC-S, CM, & IUCN. (2021). *The state of Indigenous Peoples' and Local Communities' lands and territories: A technical review of the state of Indigenous Peoples' and Local Communities' lands, their contributions to global biodiversity conservation and ecosystem services, the pressures they face, and recommendations for actions*.

Youdelis, M., Townsend, J., Bhattacharyya, J., Moola, F., & Fobister, J. B. (2021). Decolonial conservation: Establishing Indigenous Protected Areas for future generations in the face of extractive capitalism. *Journal of Political Ecology*, *28*(1). https://doi.org/10.2458/jpe.4716

Yuliani, E. L., de Jong, E. B. P., Knippenberg, L., Bakara, D. O., Salim, M. A., & Sunderland, T. (2018). Keeping the land: Indigenous communities’ struggle over land use and sustainable forest management in Kalimantan, Indonesia. *Ecology and Society*, *23*(4), art49. https://doi.org/10.5751/ES-10640-230449

Zimmerer, K. S. (2011). 'Conservation Booms' with Agricultural Growth?: Sustainability and Shifting Environmental Governance in Latin America, 1985-2008 (Mexico, Costa Rica, Brazil, Peru, Bolivia). *Latin American Research Review*, *46*(S), 82–114. https://doi.org/10.1353/lar.2011.0033

Zimmerer, K. S., Galt, R. E., & Buck, M. V. (2004). Globalization and Multi-spatial Trends in the Coverage of Protected-Area Conservation (1980–2000). *AMBIO: A Journal of the Human Environment*, *33*(8), 520–529. https://doi.org/10.1579/0044-7447-33.8.520

Zurba, M., Beazley, K., English, E., & Buchmann-Duck, J. (2019). Indigenous Protected and Conserved Areas (IPCAs), Aichi Target 11 and Canada's Pathway to Target 1: Focusing Conservation on Reconciliation. *Land*, *8*(1), 10. https://doi.org/10.3390/land8010010