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Tropical dry woodland loss occurs disproportionately in areas of highest conservation value

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Abstract

Tropical and subtropical dry woodlands are rich in biodiversity and carbon. Yet, many of these woodlands are under high deforestation pressure and remain weakly protected. Here, we assessed how deforestation dynamics relate to areas of woodland protection and to conservation priorities across the world's tropical dry woodlands. Specifically, we characterized different types of deforestation frontier from 2000 to 2020 and compared them to protected areas (PAs), Indigenous Peoples' lands and conservation areas for biodiversity, carbon and water. We found that global conservation priorities were always overrepresented in tropical dry woodlands compared to the rest of the globe (between 4% and 96% more than expected, depending on the type of conservation priority). Moreover, about 41% of all dry woodlands were characterized as deforestation frontiers, and these frontiers have been falling disproportionately in areas with important regional (i.e. tropical dry woodland) conservation assets. While deforestation frontiers were identified within all tropical dry woodland classes of woodland protection, they were lower than the average within protected areas coinciding with Indigenous Peoples' lands (23%), and within other PAs (28%). However, within PAs, deforestation frontiers have also been disproportionately affecting regional conservation assets. Many emerging deforestation frontiers were identified outside but close to PAs, highlighting a growing threat that the conserved areas of dry woodland will become isolated. Understanding how deforestation frontiers coincide with major types of current woodland protection can help target context-specific conservation policies and interventions to tropical dry woodland conservation assets (e.g. PAs in which deforestation is rampant require stronger enforcement, inactive deforestation frontiers could benefit from restoration). Our analyses also identify recurring patterns that can be used to test the transferability of governance approaches and promote learning across social–ecological contexts.

KEYWORDS

area-based conservation, conservation priorities, deforestation, protected areas, tropical and subtropical dry forests and savannas

1 | INTRODUCTION

Forest loss and degradation in the tropics and subtropics are among the most pressing sustainability issues globally. Tropical forest loss puts at risk many of the world's species and ecosystems, erodes ecosystem services that sustain local communities and global society as a whole, and fuels climate change through carbon emissions (IPBES, 2018; Parmesan et al., 2022). However, there is considerable heterogeneity in deforestation patterns and processes as well as in the conservation value of tropical and subtropical forests. Understanding where and how forest losses occur, and how these losses cause environmental impacts, is key for identifying governance strategies that maintain the ecological integrity of these forests, while taking into account the social realities of the communities that inhabit them.

Area-based conservation measures (i.e. interventions targeted over designated areas that intend to deliver effective conservation of biodiversity or other environmental assets) are a particularly important set of governance tools for safeguarding ecological systems while supporting social equity (Maxwell et al., 2020). Given limited conservation funding as well as increasing human pressure on tropical forests, there is a need to understand how different area-based conservation measures can support those biodiversity assets most in need of protection, leverage cobenefits between conservation and local communities, and foster restoration (Allan et al., 2022; Brooks et al., 2006; Bustamante et al., 2019; Jung et al., 2021). Protected areas (PAs), usually owned and governed by the state, currently cover about 17% of the terrestrial surface, and have been instrumental in achieving conservation goals (UNEP-WCMC and IUCN, 2021). In addition, many Indigenous Peoples and traditional communities govern and manage their lands in ways that are compatible with, and often actively support, biodiversity and carbon stock conservation (Forest Peoples Programme, 2020; ICCA Consortium, 2021). Some of these Indigenous Peoples' lands (IPLs) can overlap with PAs or be included in PA databases (Stevens et al., 2016). The newly established Kunming-Montreal Global Biodiversity Framework explicitly recognizes the important contributions of Indigenous Peoples and their lands and territories to biodiversity conservation (Gilbert, 2022; Stokstad, 2023). As nations move forward towards the implementation of the newly established goal of achieving 30% area-based conservation coverage by 2030 (Gilbert, 2022), identifying policy pathways to deliver equitable and effective conservation action is a critical step. This goal will only be achieved through inclusive negotiations among a variety of local and global interest groups and will have to identify context-specific conservation measures (Dawson et al., 2021). Such negotiations hinge on a sound understanding of the deforestation in regions of high cultural and environmental importance, and of the forms of woodland protection already present in these regions.

Within the tropics and subtropics, dry forests, woodlands and savannas (hereafter: tropical dry woodlands) are of particular conservation concern. Tropical dry woodlands harbour rich and unique biodiversity (Murphy et al., 2016; Pennington et al., 2018; Ribeiro et al., 2020), including unique plant assemblages and some of the last havens of megafauna in the world (Malhi et al., 2016; Pennington et al., 2006). These same ecosystems support the lives and cultures of millions of people (Schröder et al., 2021). Yet, many dry woodlands are now being replaced by other land uses, especially those driven by highly capitalized agribusiness (Buchadas, Baumann, et al., 2022; Pendrill et al., 2022). At the same time, tropical dry woodlands are often overlooked in research and policy, and by the general public (Pennington et al., 2018; Ribeiro et al., 2020; Schröder et al., 2021) and remain poorly protected, despite their high conservation value and high level of threat (Brooks et al., 2004; Hoekstra et al., 2005; Maass, 1995; Miles et al., 2006).

Much woodland loss is concentrated in deforestation frontiers, places where woodland loss is progressively expanding, typically translating into rapid or sustained processes of tree loss (Meyfroidt et al., 2018). This includes woodland loss driven by logging, settlement expansion or charcoal production, but the main driver of this loss in many dry woodlands is agriculture (Fritz et al., 2022; Hoang & Kanemoto, 2021; Pendrill et al., 2022). The diversity of drivers and social–ecological contexts under which woodland loss occurs can translate into great heterogeneity in the spatial–temporal patterns of environmental pressure. These manifest in different frontier processes, such as frontier severity (i.e. percentage of forest loss), frontier speed (i.e. expansion rate) and activeness (i.e. when frontiers are active) (Baumann et al., 2022; Buchadas, Baumann, et al., 2022). Previous work characterized and mapped typical patterns in the deforestation frontiers of dry tropical woodlands, based on identifying similar spatial–temporal patterns of forest cover and loss (Buchadas, Baumann, et al., 2022). Rampant frontiers (i.e. fast and highly severe deforestation) were detected in areas such as the Gran Chaco and Chiquitania in South America, and the tropical dry forests in Southeast Asia, while inactive frontiers where woodland loss had occurred in the past, but much woodland remained, were found in sub-Saharan Africa (Buchadas, Baumann, et al., 2022). Mapping such frontier types can help identify recurring patterns of pressure on the world's tropical dry woodlands.

Protection of forests or woodland through declaration of PAs, other effective area-based conservation (OECM), as well as support for Indigenous Peoples aiming to protect their land from deforestation often happens in response to ongoing or anticipated environmental pressure. Investigating how such areas overlap with different types of deforestation frontier could identify opportunities for more effective dry woodland protection (e.g. by providing further additional financial support for improving PA management or enforcement; Eklund et al., 2016; Janssen et al., 2018; Pacheco et al., 2021). Moreover, the importance of dry woodlands in terms of their biodiversity, carbon stocks or water is spatially heterogenous (Jung et al., 2021; Naidoo et al., 2008; Zhu et al., 2021). Thus, understanding where and how deforestation frontiers advance and the extent to which they interact with current woodland protection areas as well as conservation priorities can support developing policy responses targeted to local social–ecological conditions.

Here, we build on previous work that identified major types of deforestation frontier in tropical dry woodlands (Buchadas, Baumann, et al., 2022) and combine these data with data on (1) areas of woodland protection, (2) global conservation priorities and (3) conservation assets within tropical dry woodlands (i.e. areas with high biodiversity and carbon; hereafter: regional conservation assets). First, we investigated the contribution of tropical dry woodlands to global conservation priority areas for biodiversity, carbon stocks and water quality regulation. Second, within tropical dry woodlands, we investigated the spatial associations of deforestation frontiers, PAs, IPLs and variables describing regional conservation assets. This allowed us to assess where regional conservation assets are most threatened by deforestation frontiers and to identify situations that could require similar policy responses across the world's tropical dry woodlands. Specifically, we ask:

1. What share of global priority areas for biodiversity, carbon and water falls into the world's tropical dry woodlands?
2. How do deforestation frontiers in the world's tropical dry woodlands spatially overlap with areas of woodland protection and regional (i.e. dry woodland) conservation assets?
3. How do different frontier types relate to areas of woodland protection and regional conservation assets?

2 | METHODS

2.1 | Study area

We defined tropical dry woodlands based on maps of ecoregions (Dinerstein et al., 2017) and the Global Forest Change dataset of tree cover (Hansen et al., 2013). We build on previous work on these woodlands (Miles et al., 2006; Portillo-Quintero & Sánchez-Azofeifa, 2010; Timberlake et al., 2010) and used an inclusive definition, acknowledging different sets of definitions (Buchadas, Baumann, et al., 2022; Ocón et al., 2021). Specifically, we focused on two biomes according to the updated biome classification of Dinerstein et al. (2017): (1) tropical and subtropical dry broadleaved forests and (2) tropical and subtropical grasslands, savannas and shrublands. Within these biomes we defined woodlands as all areas with a minimum tree cover of 10% in the year 2000, based on the Global Forest Change dataset. Tree cover in this dataset refers to vegetation taller than 5 m (Hansen et al., 2013; Timberlake et al., 2010). We aggregated the initial 30-m resolution data to a $9\times9\,km^2$ grid cell, further considering all cells with more than 5% woodland cover. All cells where forests, shrublands and savannas exceeded this threshold are collectively referred to as tropical dry woodlands for the purpose of our manuscript.

Tropical dry woodlands are generally characterized by a marked dry season (typically at least 3 months), average annual rainfall from 250 to 2000 mm and often mesotrophic and dystrophic soils (Miles et al., 2006; Mooney et al., 1995; Murphy & Lugo, 1986; Timberlake et al., 2010). This typically results in a diverse vegetation structure, often with semideciduous and deciduous trees, drought-resistant shrubs or succulents and grasses (Lock, 2006; Murphy & Lugo, 1986; Pennington et al., 2006). Tropical dry woodlands have a long history of human use, but there are strong social–ecological variations both within and between woodland regions. For instance, certain woodlands support megafauna while others do not, some are used for pastoral grazing while others are not, or fire is used in some regions as a management tool but not in others. These factors impact dry forest vegetation extent and structure in distinct ways (Levis et al., 2017; Lock, 2006; Miles et al., 2006; Murphy & Lugo, 1986). These variations result, in part, from distinct agroclimatic conditions. Current land uses in tropical dry woodlands are diverse and include subsistence agriculture, shifting cultivation, pastoralism and forest resource use, including hunting, timber extraction and charcoal production (Fritz et al., 2022; Laso Bayas et al., 2022; Ryan et al., 2012). Similarly, a large body of scholarly research has documented the many material and nonmaterial cultural needs that tropical dry woodlands fulfil for Indigenous Peoples (Arenas & Scarpa, 2007; Rosero-Toro et al., 2018). Not surprisingly, the cultural identities of many Indigenous communities are intricately interwoven with the plant and animal species found in tropical dry woodlands (Camou-Guerrero et al., 2008; Suárez & Montani, 2010; Sugiyama et al., 2020). Recently, in some tropical dry woodlands, land-use change driven by industrialized agriculture for the production of soy and cattle (e.g. in Argentina, Bolivia, Brazil and Paraguay), cocoa (e.g. in Democratic Republic of the Congo), rubber (e.g. in Cambodia) and oil palm (e.g. in Mexico) has been leading to some of the highest deforestation rates worldwide (Fritz et al., 2022; Laso Bayas et al., 2022; Pacheco et al., 2021).

2.2 | Data on deforestation frontiers

Deforestation frontiers are here understood as areas where woodland loss is progressively expanding (Pacheco, 2012). To characterize and map deforestation frontiers, we used the Global Forest Change data on tree cover and annual tree cover loss for the period 2000–2020. This dataset has an overall accuracy of around 79% and 87% in the subtropics and tropics respectively (Hansen et al., 2013). Aggregating these data to a lower resolution (from 30-m to 9-km resolution in our case) can make such estimates more robust by capturing areas where deforestation is likely (Estes et al., 2018; Ozdogan & Woodcock, 2006). Our subsequent analyses were based on the aggregated, 9-km annual tree cover loss time series and reprojected for equal area coordinate system. Tree cover loss in our analysis might represent both woodland conversion to another land-cover type and or woodland degradation that does not completely change woodland to another land cover. We were not able to consider tree regeneration due to incomplete data on tree cover gain (Buchadas, Baumann, et al., 2022). To classify deforestation frontiers, we used the approach described in Buchadas, Baumann, et al. (2022) in which frontiers were defined as cells with an average annual woodland loss rate of at least 0.5% over at least 5 years. This follows the concept of frontiers as areas where woodland loss is progressively expanding (Pacheco, 2012; Rodrigues et al., 2009). For cells covered by our frontier definition, we calculated three frontier metrics that reflect different characteristics of the woodland loss process: initial woodland cover, speed of woodland loss and activeness of the frontier. Initial woodland cover refers to tree cover in the year 2000. Speed of woodland loss describes the maximum rate of change of annual woodland loss in the period 2000–2020. Both metrics were classified into two classes, high and low (initial woodland cover: high >26.33 km^2; speed of woodland loss: high >1.35 km^2/year, see Text S1, Buchadas, Baumann, et al., 2022). Our third metric, activeness, indicates when the frontier was detected, distinguishing old (from 2000 to 2015), active (before and after 2015) and emerging frontiers (from 2015 to 2020, see Text S1, Buchadas, Baumann, et al., 2022). Frontier metrics were further resampled to a 10-km grid cell by nearest neighbour for discrete data and bilinear interpolation for continuous data to match the other datasets.

2.3 | Data on areas of woodland protection

To analyse areas of woodland protection, we used datasets of PAs and IPLs, recognizing that both PAs and IPLs play critical roles in conserving woodlands and resisting highly extractive land uses (Bille Larsen et al., 2021; Fa et al., 2020; Ford et al., 2020; Sze et al., 2022; UN DESA, 2021). We nevertheless highlight the integrity and distinct nature of Indigenous People's lands in area-based conservation, recognized as a third pathway to conservation goals, beyond PAs and OECMs (CBD, 2022). This distinction is critical to supporting Indigenous Peoples' autonomy over their lands and choice in how their contributions to conservation should be recognized (ICCA Consortium, 2023). Both PAs and IPLs were estimated at a resolution of 10 × 10 km, by intersecting layers with a grid centroid to indicate presence or absence in the grid cell.

2.3.1 | Data on protected areas

For PAs, we used the World Database on Protected Areas (WDPA, UNEP-WCMC and IUCN, 2020, June 2019), which is based on data provided by government agencies and other authoritative organizations (Bingham et al., 2019). From this database, we included PAs that had their status as 'designated' or 'established' and excluded marine PAs (i.e. those that had in their designation name: 'Marine', 'Fish', 'Fisheries') and excluded PAs only available as point data. The total area included in PAs within our study area (2.9 million km^2) was spread among 3799 PAs. We applied a 10-km buffer, the minimum unit of analysis, around PAs to characterize their surroundings, because threats to areas adjacent to PA can exert pressure on PAs and these buffer areas are often targeted by conservation measures (hereafter 'Near PA') (Buřivalová et al., 2021). While we also considered OECM areas as a relevant form of conservation management, at the time of analysis there was little geospatial information available on OECMs for our study area.

2.3.2 | Data on Indigenous People's lands

For IPLs, we used the dataset compiled by Garnett et al. (2018). This dataset currently represents the most comprehensive assessment of lands where Indigenous Peoples have customary ownership, management and/or governance arrangements in place, regardless of legal recognition. The combined map, based on 127 publicly available sources, including cadastral records, participatory maps and census data, includes a total of 3.5 million km^2 that is IPLs in the tropical dry woodlands of 54 countries. The definition of Indigeneity adopted in this article follows the one in (Garnett et al., 2018) and largely aligns with that of the International Labor Organization Indigenous and

Tribal Peoples Convention 1989 (No. 169) Article 1. Importantly, this database is likely to underestimate IPLs in some countries or regions, for example because the mapping available was incomplete or the definition of Indigenous Peoples applied could have been broader (Garnett et al., 2018). We therefore caution that absences in our IPLs data do not necessarily imply that IPLs are absent outside the mapped regions, but rather indicate areas for which an Indigenous connection cannot be determined from publicly available geospatial resources. We did not use a buffer around IPLs, because in contrast to PAs, the concept of buffers is not widely used as a conservation practice for IPLs.

2.4 | Comparison to global conservation priorities

To analyse global conservation priorities, we used indicators of priority areas for biodiversity, carbon and water (BCW). To understand which share of global priority areas fell within tropical dry woodlands (RQ1), we used three conservation priorities identified by a recent global multicriteria prioritization analysis (Jung et al., 2021). First, we used priority areas based on biodiversity indicators only (hereafter referred to as B). This prioritization included both terrestrial vertebrates—from the IUCN red list of threatened species—and a representative sample (44% of all accepted taxa) of vascular plant species, the latter derived from a combination of expert-based sources and species distribution modelling. These distributions were then refined to an area of habitat (AOH), and furthermore split by biomes, to capture some intraspecific variation for different subpopulations. These were then integrated into a spatial prioritization approach using extinction-risk informed targets (for further details please see Jung et al., 2021). Second, we used priority areas derived from the consideration of the same biodiversity indicators plus indicators of above-ground, below-ground and vulnerable soil carbon (BC). Third, we used global priority areas derived by considering biodiversity, carbon stocks plus water quality regulation, the latter assessed by the potential volume of clean water by river basin, estimated by the capacity of the land uses to regulate water quality (BCW).

To analyse our global priority areas, we extracted the areas falling within our study area that correspond to the top percentile targets of 5%, 17%, 30% and 50% globally. The 5% target highlights hotspot areas. The 17% and 30% targets refer to the old (i.e. Aichi) and new (i.e. Kunming-Montreal) Convention on Biological Diversity area targets (CBD, 2022). The 50% target reflects the recently proposed Half Earth target (Locke, 2015). We then compared how much of the priority areas for global BCW fell under tropical dry woodlands against the respective priority targets (RQ1, Figure 1). If the share of global priority areas within tropical dry woodlands is greater than the target value (i.e. 5%, 17%, 30% or 50%) it would indicate that dry woodlands harbour a greater proportion of global conservation priorities than expected, highlighting their importance for conservation. Likewise, a proportion below a given target value signals underrepresentation. All conservation priority data were available at the 10-km resolution.

2.5 | Spatial association of frontiers, woodland protection and regional conservation assets

For evaluating how, within tropical dry woodlands, deforestation frontiers and areas of woodland protection spatially overlap with regional conservation assets (RQ2 and RQ3), we calculated four regional conservation assets (i.e. only for the area of tropical dry woodlands, thus capturing different aspects of conservation importance than our global priority variables). For biodiversity, we used IUCN species habitat preference data to identify woodland-dependent species. We focused on birds, reptiles, amphibians and mammals, as data on these taxa are more complete than those available for plant species (McInnes et al., 2013) and further filtered for species associated with savannas, tropical or subtropical shrublands and forest habitats. In line with Jung et al. (2020), we refined species' ranges using information on habitat associations to obtain the area of habitat in which the species could persist within IUCN ranges (Brooks et al., 2019). We then calculated three variables: (1) total richness of woodland-dependent species (S), (2) richness of threatened woodland-dependent species (S_T) and (3) the range-weighted rarity of woodland-dependent species (S_{RW}) variables. Threatened species included those classified as critically endangered, endangered and vulnerable following IUCN red list of threatened species (IUCN, 2021). Range-weighted rarity is a measure combining overall richness with the relative importance of a given grid cell for richness, by giving higher weight to range-restricted species and lower weight to wide-ranging species (Albuquerque & Beier, 2015; Tucker et al., 2012). Regarding carbon stocks (C), we used a consensus dataset of above-ground carbon based on over 20 individual maps (Spawn et al., 2020). We did not include water quality regulation in our regional analyses because regional data were unavailable. To set target areas for our regional conservation assets, we divided each variable into 100 equal sized groups, or percentiles, and extracted the top ranking 5, 17, 30 and 50 percentiles.

To determine whether deforestation frontiers (types) spatially overlap with areas of woodland protection and regional conservation assets for the world's tropical dry woodlands (RQ2, RQ3), we combined data on deforestation frontiers, PAs, IPLs and regional conservation assets, allowing us to analyse the spatial association between them (Figure 1). For assessing how deforestation frontiers overlap with areas of woodland protection and regional conservation assets (RQ 2, Figure 1), we first compared deforestation frontiers with woodland protection areas (step 1) and then compared deforestation frontiers with regional conservation assets (step 2). In step 1, we calculated the share of frontier area overlapping with woodland protection classes, both across the tropical dry woodlands and by continent. For our areas of woodland protection classification, we first classified PAs coinciding with IPLs. This included both cases, when there is an overlap of tenure regimes or IPLs are included in WDPA. Next, we mapped remaining PAs and remaining IPLs. Then, from the remaining lands, we separated areas close to PAs (i.e. within a 10 km buffer) and remaining unprotected lands.

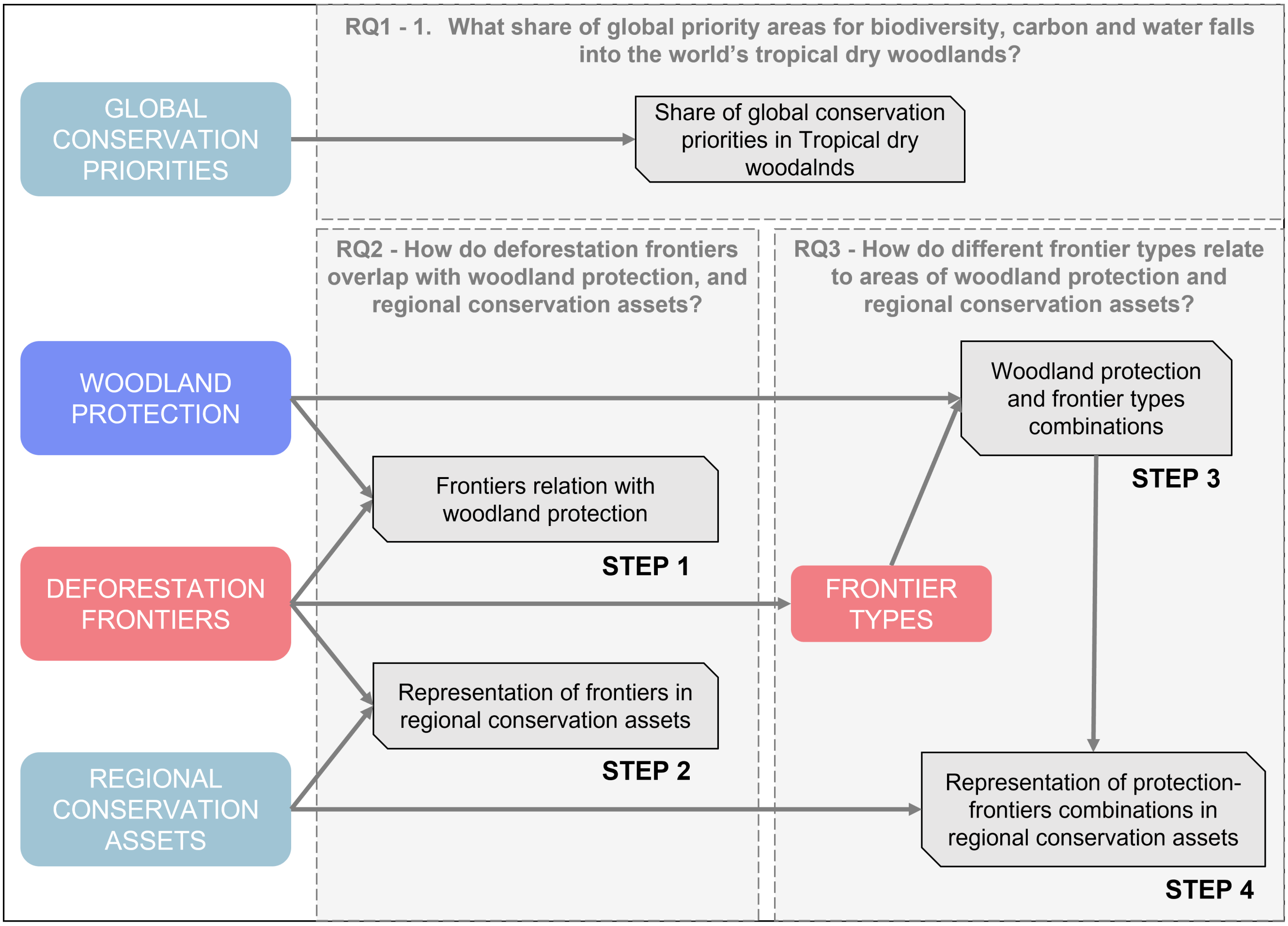

FIGURE 1 Framework to analyse the share of global conservation priorities in tropical dry woodlands (RQ1) and to assess the spatial associations between deforestation frontiers, areas of woodland protection and regional conservation assets (RQ2 and RQ3).

In step 2, we analysed how regional conservation assets are exposed to deforestation frontiers, both across the study area and within continents. For this, we used the Jacobs' index of selection (*JI*, Jacobs, 1974) to analyse whether frontier areas were over- or under-represented in regional conservation assets. We calculated:

$$JI = (oi - pi) / (oi + pi) \quad (1)$$

where *oi* is the proportion of overlap with the frontier *i* and *pi* its proportion of availability. *JI* ranges between +1 for maximum preference (overrepresentation) and −1 for maximum avoidance (underrepresentation). 'Overrepresentation', for example, refers to a situation where areas with conservation assets have a greater proportion of deforestation frontiers than would be expected based on the overall share of frontier areas in global tropical dry woodlands. For assessing how distinct types of deforestation frontier relate to areas of woodland protection and regional conservation assets (RQ 3, Figure 1), we first combined deforestation frontier types with woodland protection areas (step 3, using the same classes as in step 1) and then related those combinations to regional conservation assets (step 4). To classify major types of frontier, we used the deforestation frontier metrics on the severity and timing of woodland loss dynamics (see Figure S1). We first classified areas with fast woodland loss and a high percentage of woodland cover as rampant frontiers. We classified remaining frontiers as either emergent or old (i.e. inactive frontiers) and those that remained were classified as other, which included mostly slow-moving frontiers. In step 4, we again used the Jacob index of selection to show when combinations of frontier type and woodland protection class were over or underrepresented in regional conservation assets.

3 | RESULTS

3.1 | Representation of global priority areas within tropical dry woodlands

As defined for our study (see Figure S3), tropical dry woodlands extended over 15.76 million km^2 (~10.6% of the terrestrial surface), including large parts of Africa and South America, the south of North America, Southeast Asia and northern Australia. We found that tropical dry woodlands intersect a large share of global conservation priorities. These priority areas were overrepresented in tropical dry woodlands, highlighting their relevance for maintaining carbon stocks and water quality regulation (Figure 2). Biodiversity had the highest overrepresentation, with 9.6% of tropical dry woodlands being in the

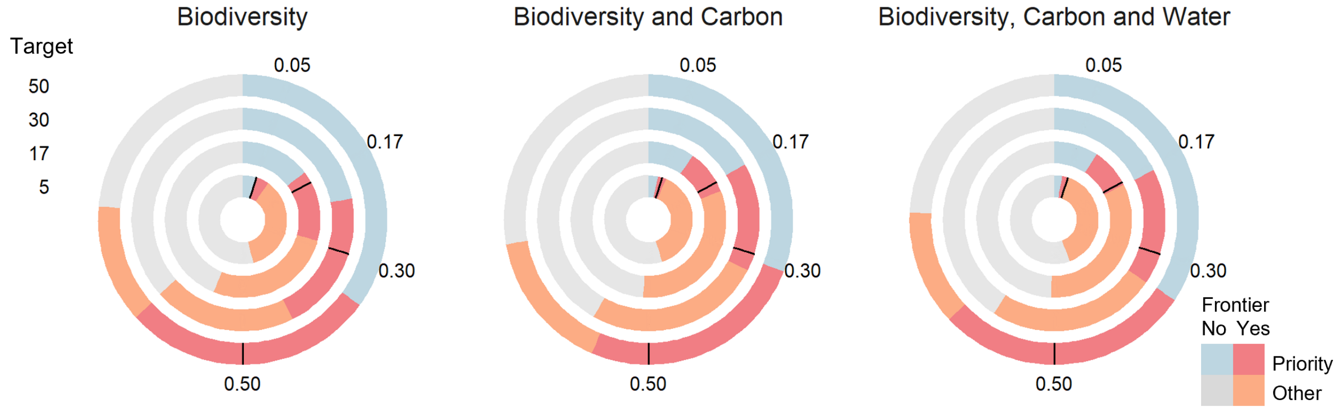

FIGURE 2 Global priority areas for three prioritization schemes (B: biodiversity, BC: biodiversity and carbon and BCW: biodiversity, carbon and water within tropical dry woodlands, from inner to outer donut are the top 5%, 17%, 30% and 50% global targets. Black lines display these percentages (i.e. the expected coverage of our study area), while the coloured circles represent the actual coverage. Blue and red parts together representing the actual share of tropical dry woodlands that are within the target value for global priorities.

top 5% of global priority areas (covering approximately 1.5 Mkm^2, so ~28% of the global 5% top priority areas). Similarly, the top 17% global priority areas accounted for 29.4% of tropical dry woodlands (~4.6 Mkm^2), the top 30% of global priority areas accounted for 42.6% (~6.7 Mkm^2) and the top 50% accounted for 63.2% (~9.9 Mkm^2). The overrepresentation was still present but lower when adding other conservation priorities. The combination of biodiversity and carbon priority areas corresponded to 6.5%, 19.0%, 32.5% and 56.4% of tropical dry woodlands for the 5%, 17%, 30% and 50% targets respectively. Finally, for the combination of carbon stocks and water quality regulation areas, the top 5%, 17%, 30% and 50% covered 5.2%, 17.6%, 34.5% and 62.9% of our study area respectively.

3.2 | Spatial associations of frontiers and woodland protection areas

Around 41% of all tropical dry woodlands fell within frontier areas (about 6.5 Mkm^2, see Supplementary analysis S1). Frontiers were more common in South America, with about half of the detected frontiers occurring there (49.8%). Remaining frontiers occurred mostly in tropical dry woodlands of Africa (39.9%), followed by Asia (5.1%), North America (3.7%) and Australia and Oceania (hereafter Australia, 1.4%). Of the woodland protection areas, we classified 4.8% of the study area as protected areas coinciding with Indigenous Peoples' lands (PA-IPLs), 11.9% as other PAs, 17.0% as other IPLs, 13.3% as near protected areas (Near PAs) and 53% as unprotected (Figure 3).

By mapping our areas of woodland protection and comparing them to deforestation frontier areas, we found that 23% of the PA-IPLs were under frontiers, corresponding to 1.1% of the total area of tropical dry woodland. Of other PAs, 28% were under frontiers (3.3% of the total area), as were 32% of remaining IPLs (5.5% of the total area), 47% of NPAs (6.3% of the total area) and 48% of other unprotected areas (25.3% of the total area, see Figure 3). Overall, PAs and IPLs were thus less likely to be classified as frontiers than were other lands. When looking at areas of woodland protection by continent, we found that in South America, the proportion of frontier areas was larger compared to areas not considered as frontier, both in close proximity to PAs and in other unprotected areas (Figure 4). In Asia and to some extent in North America, more IPLs overlapped with frontier areas than with nonfrontier areas (Figure 4; Table S1).

3.3 | Spatial representation of regional conservation assets in deforestation frontiers

We found that deforestation frontiers fall disproportionately within regional conservation assets, in particular for woodland species richness and independently of the target (Figure 5, see also Figure S6 with proportion of frontier within each target per continent). This means that the proportion of regional conservation assets identified as frontiers is greater than the proportion in tropical dry woodlands generally. In contrast, frontiers seem to encompass a disproportionately low proportion of the top 30% and top 50% areas for threatened woodland species richness. Regional trends are largely consistent with the general picture, except for South America: the overrepresentation of frontiers within regional conservation assets was higher for Asia and regional conservation assets in North America and Africa while in South America, frontiers had a disproportionately low representation in some conservation assets and in some regions (Figure 5; Figures S16 and S18).

3.4 | Spatial associations of frontier types and areas of woodland protection

To understand better how frontier types relate to classes of woodland protection, we overlapped frontier types with areas of woodland protection (Figure 3). We classified 5.4% of the tropical dry woodlands as rampant frontiers, 6.4% as emerging frontiers, 13.6% as inactive frontiers, 16.0% as other frontiers and 58.6% as nonfrontier areas (the classification of areas of woodland protection

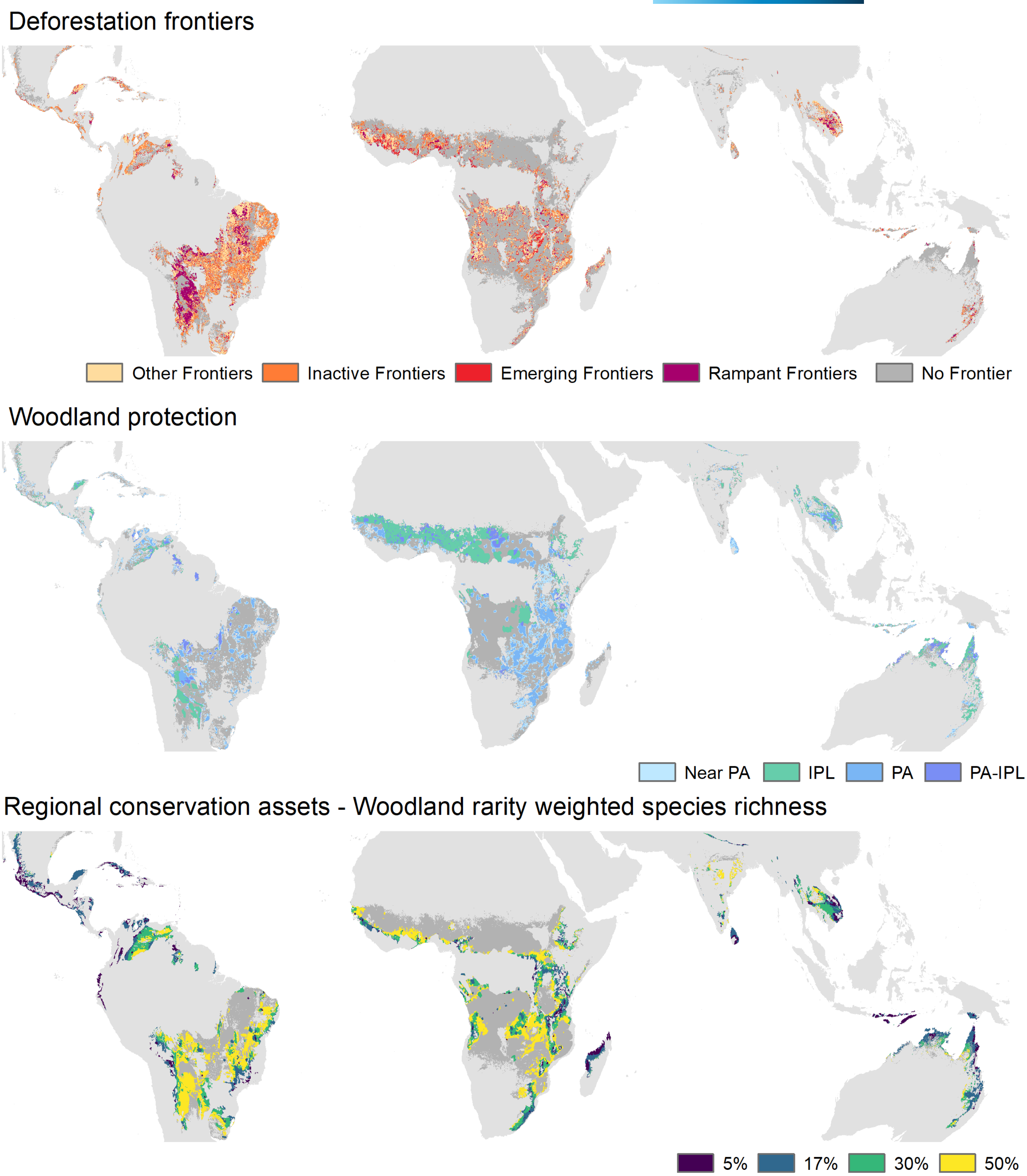

FIGURE 3 Map of deforestation frontier types (top), types of current woodland protection (middle) and example of the top-ranking cells for one regional conservation asset (bottom, here: rarity-weighted richness of woodland species). Near PAs, near protected areas; IPLs, Indigenous Peoples' lands; PAs, protected areas; PA-IPLs, protected areas coinciding with Indigenous Peoples' lands.

is above in section 'Spatial associations of frontiers and woodland protection areas'). Combining frontier types and areas of woodland protection showed that emerging frontiers were more common in areas surrounding PAs and to a lesser extent in unprotected areas outside IPLs, than in other areas of woodland protection (emerging frontiers within unprotected: 6.3%; Near PA: 8.6% of the total woodland protection area, Figure 6b). Within Indigenous Peoples' lands, either coinciding with protected areas (PA-IPLs) or not (IPLs), most frontiers were inactive (inactive frontiers—IPL: 11.8%; PA-IPL: 7.2% of the total woodland protection area, Figure 6b), while

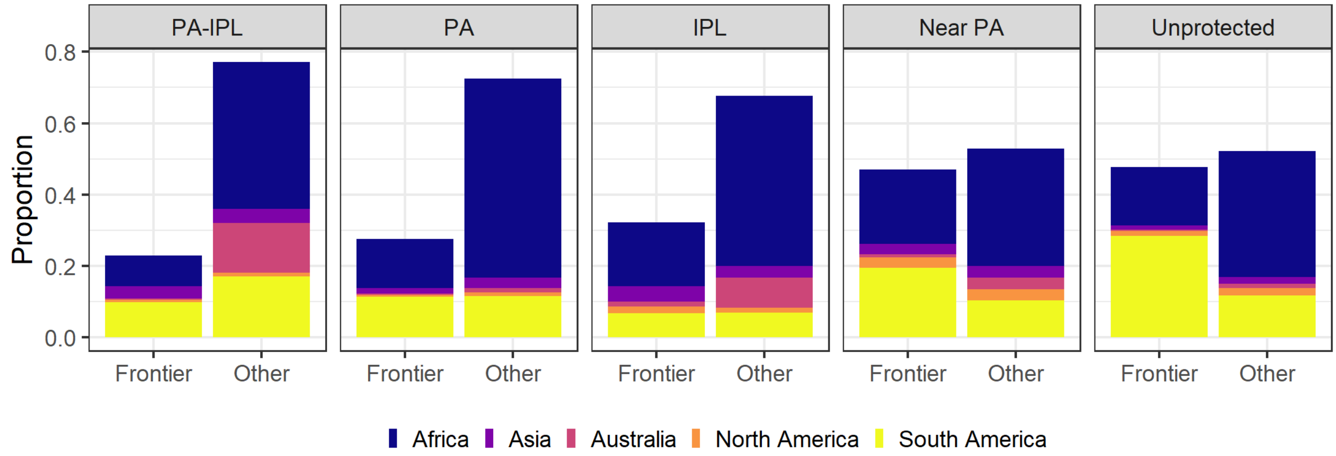

FIGURE 4 Proportion of frontier areas and other areas within classes of woodland protection. PA-IPL, protected areas coinciding with Indigenous Peoples' lands; PA, protected area; IPLs, Indigenous Peoples' lands; Near PA, near protected area and Unprotected—remaining unprotected areas, further classified by continent.

for all other categories of conservation 'other frontiers' were the most common.

For the frontiers that occurred in unprotected areas, rampant frontiers were most common in South America (e.g. Paraguay and Brazil), while emerging frontiers were most common in Africa (e.g. Guinea, Sierra Leone). When analysing proportions within continents, inactive frontiers were most common in South and North America and Australia (e.g. Brazil), while other frontiers were most common in Africa and Asia (see Table S1). Within other woodland protection classes across geographies, generally the most common were other frontiers or inactive frontiers. Noticeably, for areas near PAs we found more overlap with emerging frontiers in Africa (e.g. Guinea, Zambia) and more with rampant frontiers in South America (e.g. Paraguay). For IPLs, we found a relatively higher overlap with rampant frontiers in South America (e.g. Paraguay) and Asia (e.g. Cambodia). For PAs, we found a relatively higher overlap with emerging frontiers in Asia (e.g. Cambodia) and Africa (e.g. Guinea, Zambia) and with rampant frontiers in Australia, Asia (e.g. Cambodia) and South America (e.g. Paraguay).

3.5 | Spatial association of frontier types, woodland protection and regional conservation assets

We found that, within protected areas coinciding with Indigenous Peoples' lands (PA-IPLs), PAs and areas surrounding PAs (Near PA), frontiers were overrepresented in regional conservation assets, particularly for biodiversity (Figure 6 and see Figure S7 for proportion of frontier types within 30% target in each woodland protection class for reference). This overrepresentation poses a potential threat to those conservation assets. We also found that, overall, most regional conservation assets, with the exception of carbon, were generally underrepresented in frontier and nonfrontier areas in IPLs. Regional conservation assets were overall underrepresented in IPLs across Africa, while some of the conservation assets, that is, woodland species richness and threatened species richness, were overrepresented in Asia and North America (Figure S10).

Other frontiers often overlapped with regional conservation assets. Emerging frontiers and other frontiers were disproportionately high in areas with high carbon and woodland species richness. Emerging frontiers occurred in areas with high woodland species richness in regions like the Miombo woodlands or the Cerrado. Rampant frontiers were overrepresented in areas with high woodland species richness, for example in the Bolivian Chiquitano and the Indochina dry forests in Asia. Rampant frontiers in all woodland protection classes were also overrepresented in areas important for carbon storage, for example in the Humid Chaco, Maranhão Babaçu forests or the Sourthern Congolian forest-savanna.

4 | DISCUSSION

Tropical dry woodlands are widespread, harbour huge environmental assets and support the livelihoods and cultures of millions. Despite this, these woodlands continue to be overlooked in conservation science, policy and practice. Here, we find that these woodlands contain a disproportionate share of global priority areas for BCW regulation, underlining their importance. Yet we also show that these woodlands are threatened, with around 40% of them occurring within deforestation frontiers. We find that PAs, as well as IPLs, are comparatively less threatened by deforestation frontiers than unprotected land. However, deforestation frontiers occur disproportionately in important areas for conservation assets. This was also the case within PAs, suggesting that when deforestation frontiers expand into PAs, they do particularly so in those areas that matter the most for conservation. Among deforestation frontier types, other or inactive frontiers were the most common in all classes of the woodland protection, but we also found rampant frontiers inside some PAs, pointing to ineffective conservation and emerging frontiers nearby PAs suggesting PA isolation. Identifying

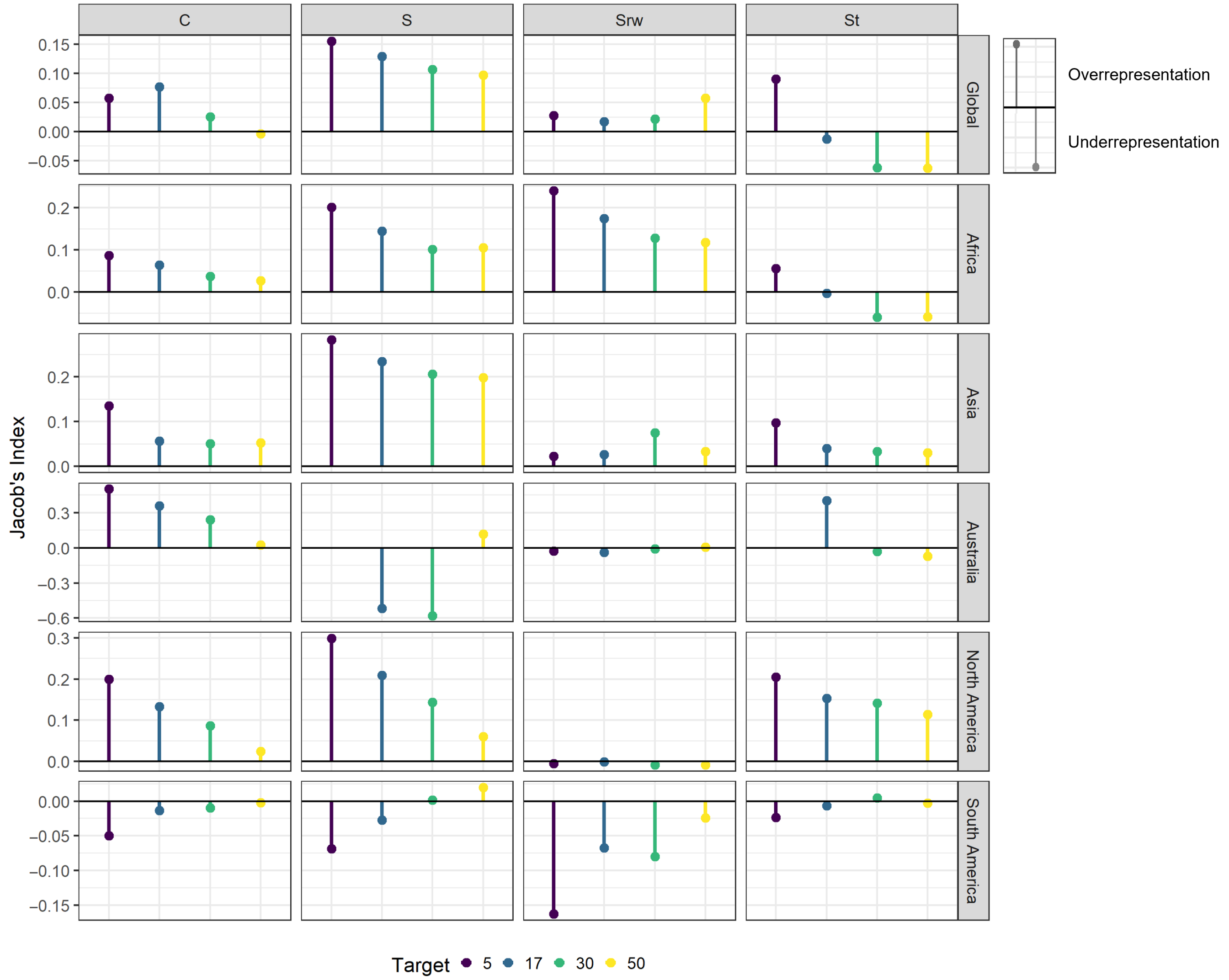

FIGURE 5 Selection index of target areas (5%, 17%, 30% and 50%) for our regional conservation assets of grid cells under deforestation frontiers for tropical dry woodlands globally and within continents. (Regional conservation assets: Carbon, S—total woodland species richness, Srw—the range-weighted rarity of woodland species and St—threatened woodland species richness. 'Overrepresentation' means that the areas within a given target area of the asset for a certain indicator within a certain region have a higher share under deforestation frontiers than the neutral expectation (i.e. the 41% overall share of frontier areas in global tropical dry woodlands).

and mapping these patterns, as we do here, enables for more targeted policy interventions and conservation management.

The conservation importance of tropical dry woodlands is starting to be recognized (Miles et al., 2006; Pennington et al., 2018; Ribeiro et al., 2020; Schröder et al., 2021) and our analysis further supports this by showing that these ecosystems are disproportionally likely to harbour global priority areas for biodiversity, carbon stocks and water quality regulation. Yet, we also uncover that many of conservation assets within these woodlands are threatened by deforestation frontiers, with potential impacts on endemic species and the climate regulation potential of these woodlands (Cardoso Da Silva & Bates, 2002; Grace et al., 2006; Romero-Muñoz et al., 2021). Geographically, there were differences in the over- and underrepresentation of frontiers, with regional conservation assets particularly affected by frontiers in Asia. This can be explained in part by the proportion of regional conservation assets on this continent and by the extensive presence of deforestation processes in some of these areas. For example, in Cambodia deforestation is a result of both economic land concessions for agro-industrial development and subsequent land disputes and land poverty, leading to the migration of smallholders towards forested more biodiverse rich uplands (Davis et al., 2015; Hayward & Diepart, 2021). Conversely, in South America, where most deforestation frontiers occur, frontiers have sometimes been underrepresented in conservation asset areas, potentially because a relatively large proportion of these assets are in PAs or IPLs (Miles et al., 2006). For example, frontiers were underrepresented in the Campos Rupestres montane savanna, in Brazil and the Guiana savanna, in Venezuela, ecoregions where a quarter to half of the land is already inside PAs or IPLs.

A key result of our analyses was that PAs and IPLs are less affected by deforestation frontiers than areas outside of them. Although our analysis is not a rigorous effectiveness assessment

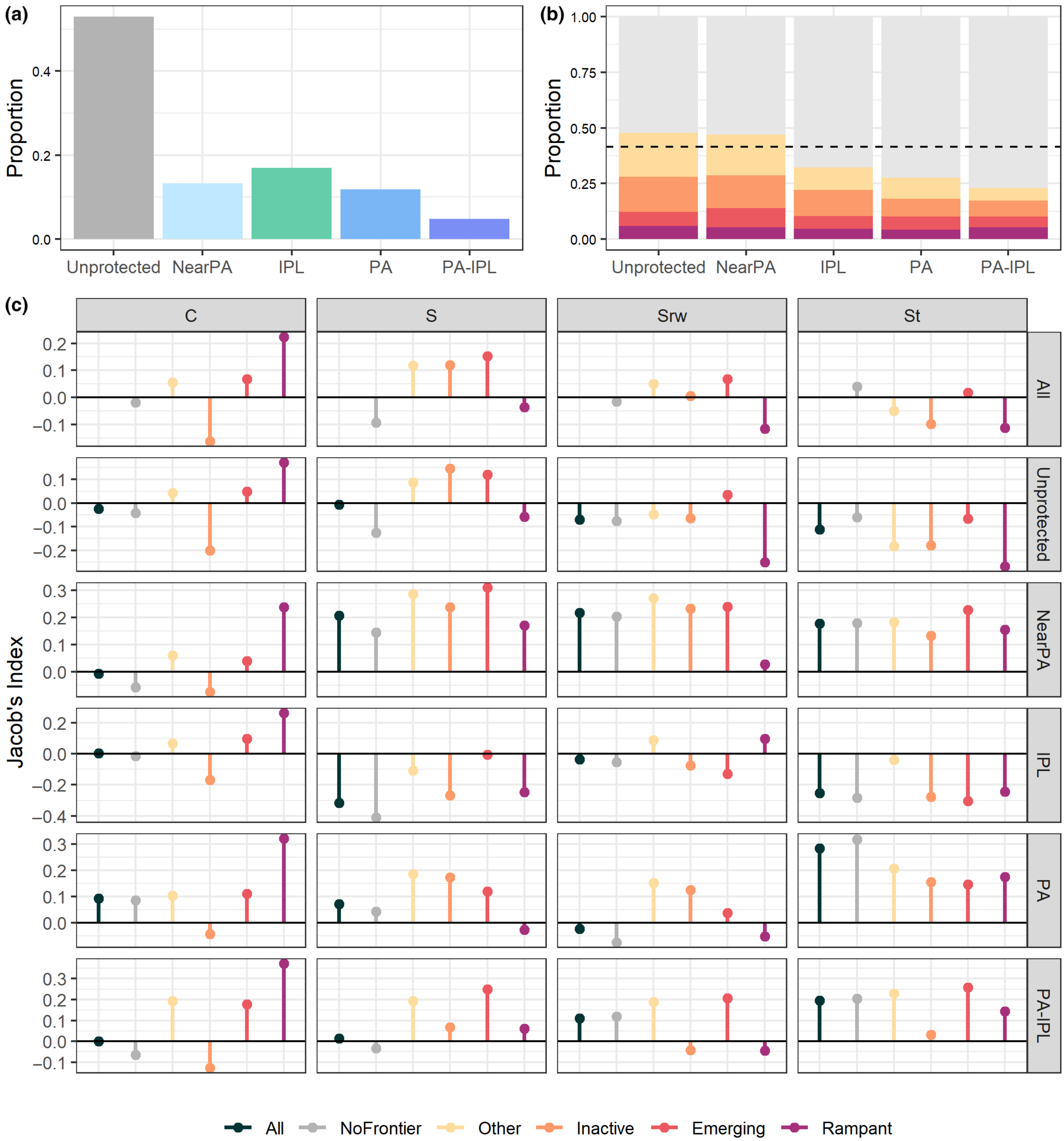

FIGURE 6 Spatial association of frontier types, classes of woodland protection and regional (i.e. dry woodland) conservation assets. (a) Proportion of each woodland protection class. (b) Frontier types within each woodland protection class; (c) Combinations of deforestation frontier types and areas of woodland protection and their relation with conservation assets (for top-30% areas). Positive values of Jacob's I signal overrepresentation and negative values underrepresentation. Regional conservation assets: Carbon, S—total woodland species richness, Srw—the range-weighted rarity of woodland species and St—threatened woodland species richness.

(Geldmann et al., 2014), our results point to a clear association of PAs and lower deforestation rates, due to PA effectiveness or because of their location (Shah et al., 2021; Wade et al., 2020; Wolf et al., 2021). Similarly, IPLs had proportionately smaller areas under deforestation frontiers than non-IPLs, irrespective of whether they coincided with PAs or not. This is consistent with scientific evidence showing that, globally, IPLs tend to be in better ecological condition than lands outside of them (Fa et al., 2020; Garnett et al., 2018; Sze et al., 2022) and emphasizes the critical importance of recognizing and upholding Indigenous Peoples' rights. Our finding is also consistent with other research in tropical dry woodlands that suggests Indigenous Peoples' stewardship can lower deforestation pressure (Pratzer et al., 2023). Violations of Indigenous Peoples' rights are frequent in tropical dry woodlands and continue to exacerbate legacies of violence, intergenerational trauma and land dispossession (Castillo et al., 2013; Correia, 2019; Jasser et al., 2021). Although at least 22% of the world's tropical dry woodlands fall inside IPLs, these lands currently receive a much smaller share of conservation funding and attention than areas protected by the state (Qin et al., 2022; Tauli-Corpuz et al., 2020). Supporting Indigenous Peoples to maintain stewardship of their lands and recognizing their historical rights to do so, therefore represents a major opportunity for enabling equitable tropical dry woodland conservation.

However, deforestation frontiers in PAs were overrepresented in areas harbouring regional conservation assets. This implies that, while deforestation in PAs is lower than elsewhere, when deforestation frontiers occur inside PAs, they may be highly detrimental for conservation. Further investigation is needed to understand why frontiers were overrepresented in regional conservation assets. Geldmann et al. (2019) found that remote PAs with lower initial human pressure have suffered more from increased human pressure, so similar processes may explain the patterns uncovered in our study. Determining the factors responsible for the disproportionately larger area of frontiers within PAs is challenging without an analysis that considers accessibility, population density, elevation or the presence of agriculture or compliance deficits (Aragão et al., 2022; Geldmann et al., 2019; Joppa & Pfaff, 2009). Additionally, we could gain further insight by specifying the types of frontiers, the potential sets of actors involved and the geographical contexts where such discrepancies occur (please see further down where we discuss frontier types in PAs).

IPLs not coinciding with PAs, however, had relatively less overlap with regional conservation assets, although this varied across geographies. For example, in Africa, conservation priorities were generally underrepresented in IPLs, whereas the opposite was true in Asia or North America. For Africa, these results need to be carefully interpreted as, first, many Indigenous People's territories are unclaimed there, as definitions of Indigeneity at the policy level remain contested and difficult to apply (Garnett et al., 2018) and, second, Indigenous Peoples have historically been disenfranchized, evicted, displaced and/or excluded from PAs, often with state-sanctioned violence (Domínguez & Luoma, 2020; Fletcher et al., 2021; Pemunta, 2019). PA-IPLs, however, had more regional conservation assets than would be expected. This might indicate either the importance of state protection or that regional conservation assets have been targeted for state protection but are also often IPLs (Stevens et al., 2016).

Although all frontier types were found in all the woodland protection classes, not all frontier types are necessarily incompatible with conservation. For example, while other or inactive frontiers could indicate encroachment by extractive industries (e.g. logging, mining), they can also be places where communities have long histories of woodland use (Jung et al., 2022; Malhi et al., 2022). The latter can balance the often strong and undesired social impacts that strictly PAs can entail (Dowie, 2011; Leberger et al., 2020). Contextualized research and policy are needed to understand how specific woodland loss dynamics might conflict with conservation goals and/or be balanced with local needs. However, rampant frontiers inside PAs and IPL are highly unlikely to be compatible with conservation and social–ecological goals. Rampant frontiers are often related to actors with the capital and power to deforest rapidly, who then attract other such actors through agglomeration economies and herd effects (Buchadas, Baumann, et al., 2022; le Polain de Waroux, 2019). Examples include the South American Gran Chaco or Cambodia's dry forests, where agribusiness expansion has been very rapid and encouraged by governments through PADDD processes including PA downgrading, or by granting economic land concessions inside PAs (Cartes & Yanosk, 2020; Ford et al., 2022). Finally, we found that areas close to PAs have many emerging frontiers. This indicates an increasing trend of PA isolation, potentially fragmenting populations of conservation concern. And eventually threatening conservation within PAs because deforestation pressure inside them increases once forests around them are lost (Buřivalová et al., 2021).

A major value of our analysis is to detect and map key combinations of frontier types and specific woodland protection classes. Previous work has shown that specific frontier patterns relate to specific actors and land uses. For example, rampant frontiers are related to commodity-driven agriculture, while fragmented frontiers, which most closely correspond to the other frontiers in this study, are associated with smallholders (Buchadas, Baumann, et al., 2022). As a result, the combinations of frontier type, woodland protection class identified here are a starting point for more context-specific conservation policies, support for Indigenous Peoples' rights and management interventions. To illustrate this, we formulated policy and management suggestions (Figure 7), including area-based, supply chain and sector specific approaches to conservation (Pacheco et al., 2021). These suggestions are meant as starting points for discussions, not as a set of prescriptions. For example, rampant frontiers in PAs might be addressed with strengthened enforcement and this could be combined with supply-chain agreements to lower deforestation pressure in regions more generally. Similarly, inactive frontiers that occur in areas of regional conservation assets could benefit from restoration efforts and, if they occur near PAs, from establishing formal buffer zones. Linking frontier type and woodland protection classes with regional conservation assets can further contextualize conservation strategies. For example, rampant frontiers such as those in the Gran Chaco, Maranhão Babaçu forests or the Southern Congolian forest-savanna, are associated with important areas for carbon stocks, so carbon credit schemes could usefully complement other forms of conservation. Nevertheless, all suggestions ultimately need to be contextualized with local realities before application, based on rights-based approaches and just processes of deliberation that prevent further marginalization of Indigenous Peoples and local communities. While conservation interventions can have positive impacts, it is important to acknowledge the potential for conflicts and rights abuses against

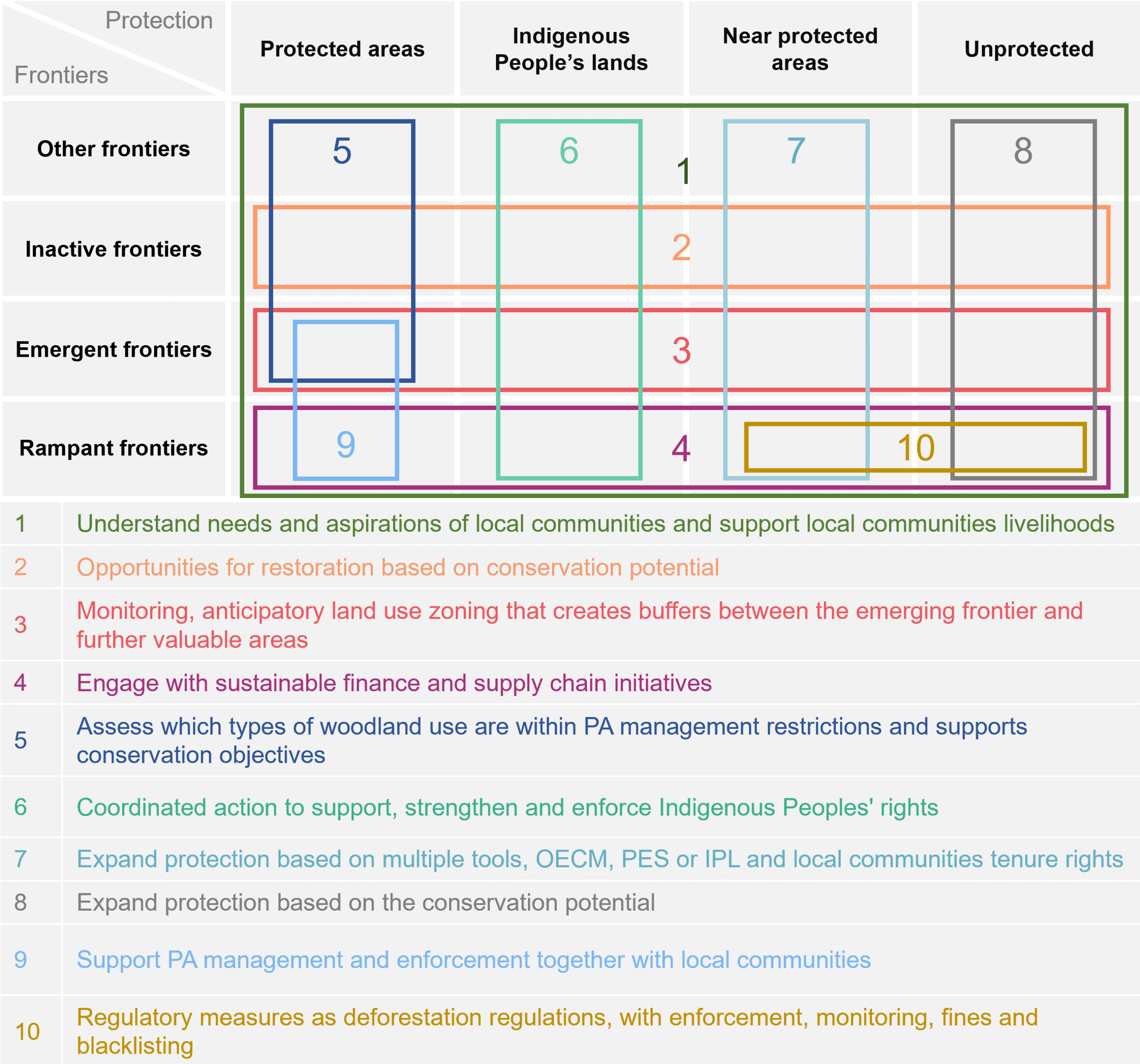

FIGURE 7 Illustration of different conservation policy and management suggestions potentially useful for specific combinations of woodland protection classes and deforestation frontier situation.

Indigenous Peoples and local communities (e.g. Moura et al., 2019; Neumann, 1997). However, there are examples where restoration, conservation programs or carbon credit schemes can work. These often led by, or in partnership with, Indigenous Peoples and local communities. To achieve this, research on successful conservation approaches has highlighted the importance of participatory and codesigning approaches for managing PAs or restoration initiatives (Moura et al., 2019; Perrotton et al., 2017). Addressing institutionalized injustices and inequalities by recognizing local traditions, knowledge, customary institutions, as well as land and resource rights of Indigenous Peoples and local communities is central to make conservation policies more legitimate, inclusive and equitable (Fernández-Llamazares et al., 2021; Robinson et al., 2021). It is also important to ensure both short-term direct benefits and long-term support for the maintenance of conservation interventions (De La Fuente & Hajjar, 2013; Reyes-García et al., 2019).

There are several reasons to be cautious about our results, although we believe none affect our conclusions. First, tree cover loss data can underestimate tree cover when trees are at low densities (Tropek et al., 2014) and some forest loss may indicate tree harvesting and not deforestation, although tree plantations are uncommon in most tropical dry woodland regions (Fagan, 2020; Fagan et al., 2022). Second, we are analysing woodland loss processes against static datasets of PAs and conservation assets. This does not allow us to acknowledge the spatial and temporal patterns of biodiversity and their conservation and that all these dimensions are a dynamic product of interacting processes (Buchadas, Qin, et al., 2022; Gardner et al., 2010). Third, our datasets and definitions of woodland used for both deforestation frontier classification and the calculation of regional conservation assets were not fully compatible thematically. For the regional conservation assets for woodland species, we used a habitat map with land cover and forest definitions that better represent species ranges (Jung et al., 2020, 2021) but these differed slightly from our definition of tropical dry woodlands in Global Forest Change dataset (Hansen et al., 2013). Fourth, we caution that overlaps with frontiers do not necessarily always translate into impact on regional conservation assets but should be seen as indicators of threat. Further ground-based work is needed to understand the impacts of different frontier types. Fifth, our regional analysis of our regional conservation assets uses areas of highest aggregated species presence to proxy conservation value. A spatial prioritization exercise including other facets of conservation value would enrich this analysis. Sixth, we consider the intrinsic value of species only, but do not capture the instrumental or relational values and significance of species for local livelihoods and this may be important when considering policy and management suggestions (Tamburini et al., 2023). Finally, we are unable to include traditional communities, those can share similar roles to Indigenous Peoples and thus remain overlooked here in their potential contributions to woodland protection (Reyes-García et al., 2022).

4.1 | Conclusions

With the new goal of 30% of global land area set for equitable and effective area-based conservation, a better understanding of where and how area-based conservation needs to expand is crucial. Tropical dry woodlands shelter 28% of the top 5% global priority areas for biodiversity conservation yet we identified 40% of the tropical dry woodlands as deforestation frontiers. Containing damaging deforestation frontiers is urgently needed if the newly agreed Kunming-Montreal goals are to be achieved. We show the importance that both IPLs and PAs had and can have in conserving these woodlands. Importantly, we explore how various conservation approaches could be applied to the different combinations of frontier type, protection status and land designation in tropical dry woodlands. Ultimately, our analysis provides a pathway to developing context-specific strategies and can facilitate learning across different social–ecological contexts.

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CONFLICT OF INTEREST STATEMENT

All the authors have read this manuscript. We have declared that no competing interests exist and consent to publication of this manuscript.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are mostly publicly available online (see methods for full source details). The data on protected areas is available at https://www.protectedplanet.net/. The data on global forest change is available at https://www.globalforestwatch.org/. The global conservation priority layers have been made openly available as part of Jung et al. (2021) at https://doi.org/10.5281/zenodo.5006332. Remaining primary data from this study are publicly available on Zenodo at 10.5281/zenodo.8033848. The map of Indigenous Peoples' Lands can only be obtained from the corresponding author on reasonable request (Garnett et al. 2018). The data are not publicly available due to privacy or ethical restrictions.

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SUPPORTING INFORMATION

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

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