Key Biodiversity Areas and Important Plant Areas could improve ecological representation in protected and conserved area networks

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Abstract

Aim: Meeting the Global Biodiversity Framework's Target 3 ("30-by-30") requires rapidly expanding protected and conserved area (PA) networks, while improving their ecological representation. We assess whether key biodiversity areas (KBAs) and important plant areas (IPAs) – sites identified using evidence-based biodiversity criteria – provide additionality across biogeographic space and species to PAs and could therefore guide more representative expansion.

Location: Twenty-eight countries with established PA, KBA and IPA networks.

Methods: We characterised ecological representation using two approaches: biogeographic space (multivariate environmental variation from bioclimatic, topographic and soil variables, using kernel density estimation) and species ranges (plants, birds, mammals, amphibians and reptiles). For PAs, KBAs and IPAs we tested for differences among networks, and quantified the additionality KBAs and IPAs could provide to existing PA networks. Finally, we tested whether land identified as a KBA or IPA before 2010 was subsequently more likely to become designated as a PA.

Results: Within countries, networks covered 35.1% (PAs), 41.8% (KBAs) and 28.1% (IPAs) of biogeographic space. The three networks occupied significantly different biogeographic space, both overall and within 82% of studied countries. KBAs and IPAs provided substantial additionality, respectively capturing 14.0% and 11.0% of biogeographic space and 66.8% and 41.0% of species missed by PAs, with threatened species more likely to be captured than non-threatened species, significantly so for IPAs. Within countries, land inside KBAs or IPAs had four-fold higher odds of subsequent PA designation than land outside these two networks.

Main Conclusions: KBAs and IPAs are strongly complementary to PAs, capturing biogeographic space and species that current networks miss. Together they offer a readily available, evidence-based foundation for guiding ecologically representative area-based conservation under Target 3 of the Global Biodiversity Framework and beyond.

Keywords: area-based conservation | biogeographic space | complementarity | ecological representation | Global Biodiversity Framework | kernel density estimate

1. Introduction

Protected and conserved areas (PAs) are the cornerstone of spatial conservation planning (Dudley et al., 2018; Jonas, Wood, et al., 2024; Lewis et al., 2019). They are the primary mechanism through which Parties to the Convention on Biological Diversity aim to achieve Target 3 of the Global Biodiversity Framework – conserving 30% of Earth for nature by 2030 (hereafter 30-by-30; see Appendix S1, Table S1.1; CBD, 2022). Critically, expansion to meet 30-by-30 must also deliver a network that is ecologically representative, covering “adequate samples of the full range of existing ecosystems, ecological processes and regions” (CBD, 2022). While PAs currently cover ~17.6% of terrestrial land and inland waters, they represent biogeographic regions and species diversity unevenly (Delso et al., 2021; UNEP-WCMC & IUCN, 2024a). Achieving 30-by-30 will therefore require the global PA network to expand at a pace far exceeding that of recent decades, while also targeting underrepresented biogeographic conditions and species (UNEP-WCMC & IUCN, 2024a).

One strategy for guiding the urgent expansion of PAs is targeting sites previously identified as important for biodiversity under related initiatives (e.g. Kullberg et al., 2019). These include Important Plant Areas (IPAs), Important Bird and Biodiversity Areas, Alliance for Zero Extinction sites, B-ranked sites, Important Fungus Areas and Prime Butterfly Areas, many of which sit under the umbrella of Key Biodiversity Areas (KBAs; Eken et al., 2004; Plumptre, Baisero, et al., 2024). These internationally recognised areas support biological processes, have high irreplaceability or ecological integrity, or are globally important for the persistence of threatened and endemic species or ecosystems (Plumptre, Baisero, et al., 2024). However, such designations do not confer formal protection. Critically, for these sites to effectively guide rapid PA expansion, they must capture diverse or underrepresented environments and species, complementary to existing PA networks; otherwise, we risk perpetuating existing biases towards specific biogeographic conditions and taxa.

Two of the largest efforts to identify sites important for biodiversity are the KBA and IPA programmes (Anderson, 2002; Darbyshire et al., 2017; Eken et al., 2004). There are currently 16,500 KBAs, including legacy KBAs that pre-date the KBA Standard and are yet to be reassessed against its criteria (BirdLife International, 2024b), and at least 2,500 published IPAs, with 76 countries having engaged with the IPA programme so far (Kor et al., 2025). Whereas protected areas tend to be established in places with low opportunity costs – such as higher elevation land with minimal agricultural value – rather than in the most important areas for biodiversity (Joppa & Pfaff, 2009; Venter et al., 2018), KBAs and IPAs use evidence-based criteria to specifically target sites with biodiversity value (Darbyshire et al., 2017; IUCN, 2016). Although KBAs are intended to be taxonomically broad, they have only been assessed comprehensively, under the KBA Standard across multiple taxonomic groups, in 11 countries to date (Plumptre et al., 2025). Furthermore, 57% of the qualifying elements of KBAs are vertebrates and less than 13% of KBAs are triggered by plants (BirdLife International, 2024b). Combining IPAs' targeted focus on endemic and threatened plants with KBAs' broader taxonomic scope may help deliver a more representative network. However, the extent to which KBAs and IPAs – individually and collectively – could improve ecological representation in PAs has yet to be assessed.

A further challenge remains that a comprehensive metric of ecological representation does not yet exist, although the coverage of biogeographic regions by protected and conserved areas is suggested as one element of the 30-by-30 coverage headline indicator (CBD, 2024; UNEP-WCMC & IUCN, 2024a). However, increased biogeographic representation is not necessarily synonymous with increased species representation (Butchart et al., 2015), because ecoregions and biomes are too coarse to comprehensively encapsulate endemism, species composition, and ecological and evolutionary processes (Visconti et al., 2019).

Here, we assess ecological representation by PAs, KBAs and IPAs in 28 countries using i) biogeographic space, which we estimate by measuring environmental variation, an approach used widely in ecology and evolution (Graham et al., 2025), and ii) species ranges for plants, birds, mammals and herptiles (reptiles and amphibians) assessed on the IUCN Red List. First, we calculate the environmental coverage of each country for its PAs, KBAs and IPAs, test whether these networks represent significantly different environmental conditions, and examine how environmental coverage scales with geographic extent across the three networks. Second, we calculate the coverage of species ranges by PAs, KBAs and IPAs, and examine whether representation by each network varies by taxa, threat status and endemism. Third, we quantify the additional coverage of biogeographic space and species ranges that KBAs and IPAs provide beyond existing PAs and examine how this additionality scales with the size of national PA networks. Finally, we evaluate evidence for KBAs and IPAs catalysing subsequent PA designations.

2. Methods

2.1. PAs, KBAs and IPAs

We identified 28 countries with PAs and established programmes to identify KBAs and IPAs, with comprehensive resulting networks – though we acknowledge that site identification is an ongoing process (Figure 1A; see Appendix S2; Table S2.2; Figure S2.1). We collated data on

PAs from the World Databases on Protected Areas and OECMs (UNEP-WCMC & IUCN, 2024b), KBAs from the World Database of Key Biodiversity Areas (BirdLife International, 2024b), and IPAs from three sources (Kor & Diazgranados, 2023; Plantlife International, 2023; RBG Kew, 2024). For India and China, which restrict spatial data for 900 and 2,960 of their protected areas respectively (UNEP-WCMC & IUCN, 2024b), we obtained protected area polygons from OpenStreetMap using the R package osmdata (Padgham et al., 2017; see Appendix S2). Many KBAs are based on Important Bird and Biodiversity Areas (IBAs), and since 93% of the total area within IBAs was also within the boundaries of KBAs (BirdLife International, 2024a) we excluded IBAs from analysis, but our KBA results remain highly applicable to them.

2.2. Environmental and species range data

We collated 31 environmental variables for climate (Fick & Hijmans, 2017), topography (EROS, 2017; Jarvis et al., 2008) and soil (Poggio et al., 2021) (see Appendix S3; Table S3.3). We calculated slope, roughness, aspect, topographic position index and terrain ruggedness index using the terrain function in the R package terra (Hijmans, 2023), and projected all variables to Mollweide equal-area at 1km resolution. We used Spearman's rank correlation coefficient to assess multicollinearity across the 28 countries' combined geographic space, removing highly rank-correlated ($\rho > 0.7$) variables and any remaining variables with a high variance inflation factor (>5), resulting in 13 retained environmental variables (Table S3.3).

We obtained range data from the IUCN Red List in April 2024 for 2,527 threatened (Critically Endangered, Endangered and Vulnerable) and 12,480 non-threatened (Near Threatened and Least Concern) plant, bird, mammal and herptile species listed under at least one of the studied countries (IUCN, 2024; Table S3.4). Following the KBA guidelines (KBA Standards and Appeals Committee, 2022), we filtered the dataset to only retain range polygons coded as extant or probably extant and native, reintroduced or assisted colonisation, leaving 2,473 threatened and 12,413 non-threatened species. As the IUCN Red List does not yet contain comprehensive spatial polygon data for plant species – approximately 86% of red-listed plant species in our studied countries did not have range maps – we generated ranges for an additional 2,394 threatened and 10,102 non-threatened plant species using the R package rCAT (Moat et al., 2020; see Appendix S3). Our final dataset thus contained ranges for 4,867 threatened and 22,515 non-threatened species.

2.3. Biogeographic representation by PAs, KBAs and IPAs

We first calculated the geographic coverage of each country by PAs, KBAs and IPAs as the proportion of its total land area overlapping with each of these networks. We then tested the performance of PAs, KBAs and IPAs in representing the overall variation in a country's bioclimatic, topographic and pedologic conditions – what we refer to hereafter as a country's "background" biogeographic space. For each country, we extracted the retained environmental variables from a geographically stratified sample of points within its national boundary (see Appendix S4). We then conducted principal component analysis to characterise the country's background biogeographic space (Figures S4.2, S4.3) and determined which of the geographically sampled and corresponding principal component analysis points fell within PA, KBA or IPA networks. Using the first three principal components (PC1-PC3), which together explained two-thirds of the variation in each country's environmental data on average, we applied two methods to characterise biogeographic representation. For both methods, we defined the full extent of a country's background biogeographic space as the volume of a convex hull enclosing all background points. Both methods therefore measure each network against the same total volume, differing only in how each network's occupied region is delimited, making them directly comparable.

First, we calculated minimum convex polygons (MCP) for each network in three-dimensional principal component space using the R package geometry (Habel et al., 2025). We fitted 99% MCPs, excluding the 1% of points furthest from the network's centroid, to avoid inflating environmental coverage based on outliers. We then calculated the proportion of the background hull occupied by each network's MCP by Monte Carlo integration, drawing 250,000 uniformly distributed points within the bounding box of each country's background hull, retaining those falling inside the hull, and calculating the proportion that also fell within the network's MCP.

Second, we fitted a kernel density estimate (KDE) to each network's points in the same three-dimensional space using the R package ks (Duong, 2007). We used a fixed bandwidth ($h = 0.5$) for every country and network to maintain comparability and conducted sensitivity analysis at narrower ($h = 0.4$) and wider ($h = 0.6$) bandwidths (see Appendix S4). Absolute coverage varied accordingly, but the ranking of countries was almost unaffected by bandwidth (Spearman ρ: 0.92 – 0.99 across six coverage measures; Figure S4.4), as was the ordering of networks within countries, which changed only where networks were closely matched. We therefore report results at $h = 0.5$, noting that absolute coverage values, but not the comparisons drawn from them, depend on this choice. From each density estimate we delimited the 99% highest density region (HDR; Hyndman, 1996), the smallest-volume region containing 99% of the network's estimated density, as:

$$R_i = \{x: \hat{f}_i(x) \geq f_\alpha\}$$

where $\hat{f}_i$ is the estimated kernel density for network i and f_α (with α = 0.01) is the largest constant such that $\int_{R_i} \hat{f}_i(x)\, dx \geq 1 - \alpha$ (Hyndman, 1996). We calculated biogeographic coverage as the volume of the HDR within the background convex hull divided by the volume of that hull, evaluated by counting cells on a 120^3 grid in three-dimensional principal component space spanning the background points (see Appendix S4 for sensitivity analysis).

HDR and MCP produced significantly and positively rank-correlated coverage estimates across countries (Spearman ρ: 0.55–0.92, all p < 0.01; see Appendix S4, Table S4.5), indicating the ordering of countries was largely robust to method choice. We use the more conservative, density-driven HDR hereafter and report MCP findings in the supporting information. Because the background convex hull may include environmental conditions a country does not itself contain, we include the proportion of the hull occupied by the background as a covariate in all cross-country regressions (see Methods below). To test for significant differences in biogeographic coverage across networks, we conducted PERMANOVA and post-hoc pairwise adonis tests with Bonferroni correction, both nationally and pooled across all countries, using spatially thinned or stratified subsamples to avoid pseudoreplication (see Appendix S4)

To determine which variables characterised environmental differentiation between PAs, KBAs and IPAs, we plotted the overall density distribution and median values of each variable for each network using the raw environmental extractions from all geographically stratified points and applied Kruskal-Wallis and post-hoc Dunn's tests with Bonferroni correction. To examine how environmental coverage varies with the geographic extent of each network, we fitted log-log linear regressions of environmental coverage against geographic coverage, both with and without the hull-occupancy covariate. We then tested whether slopes differed across networks using an interaction term between log geographic coverage and network, and compared the fitted slopes using estimated marginal trends in the R package lsmeans (Lenth, 2016).

2.4. Species range representation by PAs, KBAs and IPAs

To quantify spatial overlaps, we intersected each species' range with each country's PA, KBA and IPA polygons in a Mollweide equal-area projection, measuring the area (km^2) of range falling within each network. For each species in each country, we calculated the proportion of its national range covered by each network. We repeated this using pairwise and three-way unions of the network polygons within each country. Each species was characterised by its IUCN Red List category and whether it was endemic. Species were classed as threatened or not threatened. In total, 4,598 threatened and 21,615 non-threatened species had ranges overlapping at least one of the studied countries (Figure 1B).

We assessed species range representation in two ways. First, we calculated the mean and standard error of overlap percentages for each network within each taxon, threat status and endemism class, and plotted these values. Second, to account for differences in the geographic extent of PAs, KBAs and IPAs, we divided each species' overlap percentage by the total area of the corresponding network in that country, giving an area-standardised measure of representation.

To test whether PAs, KBAs and IPAs cover species' national ranges differently, we conducted Friedman tests, treating each species-country combination as a block and the three networks as repeated measures within it (51,481 complete blocks). For all significant results, we conducted pairwise Wilcoxon signed-rank tests, Bonferroni-corrected across the three network pairs. We ran this on both raw coverage and per km^2 of network, and repeated it within each taxon, threat status and endemism class to see whether differences were consistent across groups of species.

2.5. Additionality of KBAs and IPAs beyond PAs

To determine whether KBAs and IPAs provide additional environmental coverage beyond PAs, we quantified the proportion of each country's background biogeographic space falling within the KBA or IPA highest density regions but outside that of PAs. We summarised additionality across the 28 countries by its mean, standard deviation and median, and tested whether KBA and IPA additionality differed using a paired Wilcoxon signed-rank test. To assess how environmental additionality scales with the geographic extent of PAs, we regressed each network's additionality against the log-transformed geographic coverage of PAs using ordinary least squares, fitting regressions separately for KBAs and IPAs, both with and without the hull-occupancy covariate, and reporting slope coefficients, R^2 and p-values.

To quantify species additionality – species present in KBAs and/or IPAs but not in PAs – we identified, for each country, species whose national range had no spatial overlap with any PA but whose overlap with KBAs or IPAs in that country was non-zero. We also repeated this across all countries, identifying species with no PA overlap in any studied country. These additional species were characterised by taxonomic group, threat status and endemism and partitioned into those covered by KBAs only, IPAs only and both networks. To assess how species additionality scales with the size of national PA networks, we expressed additionality as the proportion of species missed by PAs that were additionally covered by each network and regressed this against the log-transformed geographic coverage of PAs using the same ordinary least squares approach.

2.6. KBAs and IPAs as catalysts for PA designation

Finally, we tested whether land identified as a KBA or IPA by 2010 was subsequently more likely to be designated as a PA than land outside these two networks. The 2010 threshold was chosen to provide a sufficiently large window for subsequent designation, while also ensuring well-established networks of KBAs and IPAs against which to test the relationship. For each country, networks were rasterised to a 1km grid in a Mollweide equal-area projection. We excluded cells that already fell within PAs in 2010, as these could not subsequently be newly designated, along with cells whose KBA/IPA status in 2010 was unclear, and classified the remainder by their KBA/IPA status in 2010 (inside/outside) and PA status in 2024 (inside/outside), yielding a 2x2 contingency table per country (Table S4.6). For each country, and for all studied countries pooled, we conducted chi-squared tests and calculated odds ratios to quantify the magnitude of any association and Cramer's V to measure effect size (see Appendix S4).

Because neighbouring cells are non-independent, tests performed on pooled cells may artificially inflate the significance of any association. We therefore treated each country as a single independent observation, fitting a binomial generalised linear mixed model to the country-level contingency tables, allowing the effect of KBA/IPA status to vary between countries. We thus report the odds ratio for a typical country and the variation in this effect across countries. China and India were excluded from this analysis because their PA data from OpenStreetMap did not contain information on the year of designation (see Appendix S2).

3. Results

3.1. Biogeographic representation by PAs, KBAs and IPAs

KBAs captured the largest proportion of countries' total biogeographic variation, averaging 41.8 ± 15.5% (mean ± standard deviation), followed by PAs (35.1 ± 19.7%) and IPAs (28.1 ± 18.4%) (Figure 2A; see Appendix S5, Figure S5.7). Environmental coverage by all three networks combined averaged 52.0 ± 14.6%, ranging from 21.8% in Libya to 76.4% in the United Kingdom (Table 1; Figure 2B; Table S5.7). Representation of biogeographic space across all studied countries differed significantly between PAs, KBAs and IPAs ($F = 61.7$, $p = 0.001$). At the national level, environmental coverage differed significantly between PAs, KBAs and IPAs in 23 (82.1%) countries. Post-hoc comparisons revealed significant differences between PAs and KBAs in 16 countries, PAs and IPAs in 17 countries and KBAs and IPAs in 18 countries (Table S5.8).

Of the 13 retained environmental variables, 11 (not aspect or topographic position index) showed significant differences in their distribution across the three networks (Table S5.9). Both KBAs and IPAs tended to have greater elevation, slope, precipitation seasonality and lower mean diurnal temperature range, mean temperature of the wettest and driest quarters, compared to PAs (Figure 3A; Figure S5.8). KBAs and IPAs differed for isothermality. Generally, the magnitude of difference between variables was greater between IPAs and PAs than between KBAs and PAs.

Coverage of biogeographic space by PAs, KBAs and IPAs each scaled positively with geographic coverage (PAs: $\beta = 0.47$, $p < 0.001$; KBAs: $\beta = 0.27$, $p = 0.005$; IPAs: $\beta = 0.42$, $p < 0.001$; Figure 3B). However, all three slopes were substantially lower than one, indicating that environmental coverage increases more slowly than geographic coverage. Slopes did not differ significantly among networks ($F = 0.91$, $p = 0.405$), and no pairwise comparison was significant (PAs vs KBAs: $\Delta\beta = 0.20$, $p = 0.372$; PAs vs IPAs: $\Delta\beta = 0.05$, $p = 0.870$; KBAs vs IPAs: $\Delta\beta = -0.15$, $p = 0.576$; Table S5.10). Including the proportion of the hull occupied by the background as a covariate left the PA and IPA slopes nearly unchanged, but reduced the KBA slope to 0.16 ($p = 0.131$).

3.2. Species range representation by PAs, KBAs and IPAs

Across all taxa, PAs covered an average (mean ± standard deviation) of 23.1 ± 20.2% of species' ranges, KBAs 21.2 ± 19.4% and IPAs 8.6 ± 14.8% (Table 2; see Appendix S6, Table S6.11). Together, the three networks covered an average of 35.0 ± 22.0% of species' ranges, 11.9% more than PAs alone. PAs, KBAs and IPAs covered, on average, a higher proportion of threatened or endemic species' ranges than non-threatened species ranges, for almost all taxonomic groups (Figure 4A). KBAs covered a lower average proportion of countries' land area than PAs (17.1% vs 21.3%), yet captured a higher mean proportion of threatened species' ranges in all taxonomic groups, most markedly for herptiles (+10.4%; plants, +2.1%; mammals, +1.7%; birds, +0.1%). Amongst endemics, the advantage was large for herptiles (+7.2%) and mammals (+8.3%), but negligible for plants (+0.2%) and reversed for birds (-1.7%).

IPAs, as the smallest network in geographic space, incorporated the lowest proportion of species' ranges for all taxa and combinations of threat and endemic status. However, threatened plant species (mean ± standard error: 13.9 ± 0.4%) were the best represented threatened taxonomic group in IPAs (birds: 8.4 ± 0.6%; herptiles: 11.3 ± 0.7%; mammals: 11.0 ± 0.7%). By contrast, PAs were slightly less representative of threatened plant species (24.3 ± 0.5%) than threatened birds (24.6 ± 0.9%), herptiles (27.0 ± 0.9%) and mammals (27.8 ± 1.1%). Raw coverage differed significantly across all three networks (Friedman test: X^2 = 29,719.0, df = 2, $p < 0.001$), and all pairwise differences remained significant after Bonferroni correction (Wilcoxon signed-rank, all $p < 0.01$; Table S6.12).

When accounting for differences in the geographic extent of each network, IPAs achieved the highest mean coverage per unit area for all four threatened taxonomic groups. This was most pronounced for threatened plants, which were better represented per 1,000 km^2 of network in IPAs (mean ± standard error: 6.54 ± 1.18% per 1,000 km^2) than PAs (5.59 ± 1.43%) or KBAs (6.17 ± 1.31%). IPAs also had the highest mean per-area coverage of endemic plant species (0.97 ± 0.09% per 1,000 km^2; PAs: 0.61 ± 0.10%; KBAs: 0.71 ± 0.06%). Size-adjusted coverage also differed significantly across all three networks (Friedman test: X^2 = 2,627.3, df = 2, $p < 0.001$), and almost all pairwise differences remained significant after Bonferroni correction (Table S6.12).

3.3. Additionality of KBAs and IPAs beyond PAs

Across all 28 countries, KBAs covered an average of 14.0 ± 15.6% (mean ± standard deviation) of background biogeographic space not captured by PAs, while IPAs covered 11.0 ± 14.6% (Figure 2A; see Appendix S7, Table S7.13). KBAs provided significantly greater environmental additionality than IPAs (Wilcoxon signed-rank: V = 300, $p < 0.05$), exceeding them in 20 of the 28 countries. Additionality approached zero in a small number of countries – lowest for KBAs in Montenegro (0.2%) and for IPAs in Bolivia (0.0%) – reflecting countries where networks were largely co-located with existing PAs in biogeographic space. Türkiye showed the highest additionality for both networks (KBAs 61.2%, IPAs 60.0%).

In total, 1,143 species' ranges did not overlap with PAs in any of the 28 countries. Of these, 66.8% (763) were captured by KBAs and 41.0% (469) by IPAs (Table S7.13). KBAs were particularly effective at additionally capturing birds (83.7% of bird species missed by PAs), while IPAs were proportionally most effective for additional plant species (49.3%). The same pattern was evident when additionality was measured as range area rather than species counts. Across all species, PAs covered a mean of 23.1% of national ranges, rising to 32.4% once KBAs were included, and 35.0% with IPAs also added (Figure 4B). Threatened species showed slightly higher rates of additionality than non-threatened species for both KBAs (67.3% vs 65.8%) and IPAs (43.3% vs 36.9%). Consistent with this, logistic regression controlling for taxon and endemism indicated that threatened species were significantly more likely to be additionally captured by KBAs (odds ratio, OR = 1.34, 95% CI: 1.01-1.79, $p < 0.05$) and IPAs (OR = 1.40, 95% CI: 1.06-1.85, $p < 0.05$; Table S7.14).

Both environmental and species additionality declined with increasing PA geographic coverage, though the strength and significance of this relationship varied by network and metric. Environmental additionality declined significantly for both KBAs ($\beta = -5.71$, $R^2 = 0.33$, $p < 0.01$) and IPAs ($\beta = -5.53$, $R^2 = 0.35$, $p < 0.001$), whereas for species additionality the decline was significant for IPAs ($\beta = -10.91$, $R^2 = 0.29$, $p < 0.01$) but not KBAs ($\beta = -4.89$, $R^2 = 0.08$, $p = 0.152$) (Figure S7.9). R^2 values were moderate throughout, except for KBA species additionality, indicating that PA geographic coverage explains some but far from all of the variation in additionality. Including background hull fill as a covariate raised adjusted R^2 from 0.30 to 0.47 for KBAs and from 0.32 to 0.42 for IPAs, suggesting that some of the unexplained cross-country variation reflects the geometry of this hull in biogeographic space, rather than additionality itself.

The probability that a PA-missed species was additionally captured also differed among taxonomic groups. Considering the species missed by PAs across all 28 studied countries, logistic regression revealed significant effects for some taxa for both networks. Relative to birds (the reference group), herptiles and plants were significantly less likely to be additionally captured by KBAs (herptiles: OR = 0.43, 95% CI: 0.18-0.92, $p = 0.040$; plants: OR = 0.37, 95% CI: 0.16-0.78, $p = 0.015$). For IPAs, by contrast, plants were significantly more likely than birds to be captured (OR = 2.23, 95% CI: 1.20-4.31 $p = 0.013$; Table S7.14).

3.4. KBAs and IPAs as catalysts for PA designation

From 2010 to 2024, land within KBAs/IPAs was significantly more likely to be designated as a PA than land outside these two networks. The odds of a KBA/IPA cell becoming a PA were approximately 4.1 times higher than the odds for a cell outside these networks (95% CI: 1.56-10.91, $p < 0.01$). Country-level odds ratios varied from 0.05 in Bhutan to 508.9 in Türkiye (median = 4.23), and was >1 in 15 of 24 countries (excluding India and China, for which we did not have information on PA designation years, and Libya and British Virgin Islands, where no cell was newly designated as a PA after 2010) (Figure 5). Chi-squared tests showed significant associations in 20 countries (Table S8.15), but these were only positive in 13, with land in KBAs/IPAs significantly less likely to be designated than land outside those networks in seven countries. Effect sizes (Cramer’s V) were small across all countries (0.11 ± 0.15, mean ± standard deviation) and for the pooled table across countries (0.005), partly because only 3.70% of cells became PAs across the 14-year survey period. Pooled across all countries, however, the association was much weaker (OR = 1.13, 95% CI: 1.11-1.15), because KBA/IPA land tends to be concentrated in countries that designated limited new PAs, while new designations were mostly in countries with small KBA/IPA networks.

4. Discussion

To meet the ambitious area-based conservation targets of the Kunming-Montreal Global Biodiversity Framework (GBF; CBD, 2022), countries must urgently determine how and where to expand their protected and conserved area networks. Critically, Target 3 requires this expansion be ecologically representative, yet PAs have often been placed where socioeconomic opportunity costs are low rather than where biodiversity value is highest (Joppa & Pfaff, 2009; Venter et al., 2018). We show that KBAs and IPAs are complementary rather than redundant additions to PA networks, capturing biogeographical conditions and species that are underrepresented. These sites therefore provide an evidence base for steering expansion towards greater ecological representation, if socioeconomic challenges can be equitably mitigated.

4.1. PAs, KBAs and IPAs represent different biogeographic space

We find that PAs, KBAs and IPAs differ significantly in the biogeographic space they cover, both globally (across our dataset) and within 82% of assessed countries. Specifically, we found that KBAs and IPAs occupy higher, steeper, more seasonal environments, with less variable and cooler temperature regimes than PAs, and that IPAs diverged most strongly. This likely reflects the biogeographical conditions that drive patterns of endemism across different taxa. For example, the association between endemism and increasing elevation may be stronger in plants, with reduced dispersal capabilities than in birds (Darbyshire et al., 2017; Irl et al., 2015; Noroozi et al., 2018).

Environmental coverage increased significantly for all three networks, but in every case the relationship was sub-proportional, meaning that each additional unit of land covered contributed progressively less new biogeographic space. The rate of this scaling did not differ significantly between networks, so we found no evidence that any one network captures new environmental conditions any more efficiently than the others as it expands.

4.2. Coverage of underrepresented taxa by KBAs and IPAs

Over 70% of vascular plant species have not yet had their conservation status formally assessed (Bachman et al., 2024), and three-quarters of undescribed plant species may already be threatened with extinction (Brown et al., 2023). This gap is reflected in the KBA network, in which fewer than 13% of KBAs are triggered by plant species and the majority of qualifying elements are vertebrates (BirdLife International, 2024b). While comprehensive, multi-taxon assessment is underway, having been completed in 11 countries (Plumptre et al., 2025), IPAs, whose criteria are explicitly designed for threatened and endemic plant species (Darbyshire et al., 2017), offer a targeted means of closing this gap. However, the programme is much more limited in geographic scope than KBAs: across our 28 studied countries, KBAs covered over double the land area of IPAs (Table S2.2).

Consistent with this, PAs, KBAs and IPAs differed significantly in their coverage of species ranges across taxa. In absolute terms, PAs and KBAs covered a greater proportion of species' ranges than IPAs, including plant ranges, reflecting their broader geographic extent. After accounting for network size this ordering reversed, with IPAs achieving the highest mean coverage per unit area of plant ranges in all three status groups. This apparent efficiency should be interpreted cautiously, given the sub-proportional scaling of coverage with area noted above. A clearer plant-specific signal emerges when comparing taxa within IPAs, where network size is held constant and threatened plants were the best represented group. This is to be expected, given that IPA criteria are tailored towards features of plant diversity. This has conservation significance, implying that even modest, well-targeted additions to PA networks could improve the representation of threatened and endemic plant species considerably. This aligns with previous work showing that, by capturing sites rich in threatened, endemic and range-restricted plant species, IPAs can improve plant representation in conservation planning, particularly where existing overlap between IPAs and PAs is low (Richards et al., 2023). While our analyses illustrate the potential for KBAs and IPAs to improve biogeographic and species representation, we acknowledge that there remain many other axes of variation, including functional and phylogenetic diversity, as well as connectivity, which capture alternative facets of biodiversity explicitly mandated in GBF Target 3 (CBD, 2022).

Per unit area, KBAs and IPAs captured threatened species more effectively than PAs across most taxonomic groups. This is consistent with Criterion A of the KBA Standard and IPA programme being explicitly designed to identify sites supporting threatened biodiversity (Darbyshire et al., 2017; IUCN, 2016). It also supports evidence that KBAs represent threatened and near-threatened species better than expected by chance (Lansley et al., 2025). The direction of this relationship is not straightforward, however. Species may be more likely to be threatened precisely because they fall outside PA networks, where formal protection and *in situ* management are lacking. Either interpretation strengthens the case for integrating KBAs and IPAs into PA networks.

4.3. Additionality provided by KBAs and IPAs

We show that both KBA and IPA networks provided additionality, capturing biogeographic conditions and species' ranges that fell entirely outside PAs. Additionality was particularly evident for species of conservation concern. Threatened species were more likely than non-threatened species to be additionally captured by both networks, consistent with the explicit targeting of threatened biodiversity under their criteria (Darbyshire et al., 2017; IUCN, 2016). Additionality declined significantly with national PA coverage for three of the four relationships examined (environmental and species additionality, KBAs and IPAs). Such declines are an expected consequence of network complementarity, as each network can add less where more is already protected. This carries a direct planning implication: the representational return on incorporating KBAs and IPAs is likely greatest in countries with the least existing PA coverage, which are also those requiring the largest expansion to meet 30-by-30. PA extent alone, however, explained less than roughly a third of the variation in additionality ($R^2 < 0.35$), indicating that country-specific factors, such as the size and spatial configuration of KBA and IPA networks, may matter substantially. Furthermore, low additionality should not be equated with redundancy – a network can add little new biogeographic space simply because it occupies environments already represented within existing PAs, while still extending conservation attention and monitoring to distinct sites and the species they support.

4.4. KBAs and IPAs as building blocks for PA designation

Given that KBA and IPA identification is underpinned by evidence, we tested whether they have already guided PA expansion. Examining how protection status changed between 2010 and 2024, we found that previously unprotected cells inside KBAs or IPAs were typically 4.1 times more likely to a PA compared to cells outside them. This is encouraging given that KBA coverage by PAs is itself an official indicator for the sustainable development goals (15.1.2) and GBF Target 3 (CBD, 2022; United Nations, 2015). Yet, a substantial share of KBAs remain incompletely covered and approximately one-third of KBAs fall entirely outside PAs (UNEP-WCMC & IUCN, 2024a). This association, while statistically significant, should not be over-interpreted as evidence of catalysis. KBAs and IPAs are identified on the basis of biodiversity value, which is also among the broader criteria guiding PA designation, so the association may partly reflect shared selection criteria rather than causal influence. The absolute probability of conversion remained low even inside KBAs and IPAs, and the overall effect size was weak (mean Cramer's $V = 0.11 \pm 0.15$). The strength and even the direction of the relationship varied markedly, with land inside KBAs/IPAs significantly less likely to be designated in seven of the 24 countries assessed, indicating that national policy context, rather than KBA or IPA status *per se*, governs whether PA designation follows. In parts of Eastern Europe, for instance, IPA protection was driven largely by incorporation into the Natura 2000 network as countries joined the EU, rather than by IPAs catalysing protection (Kor et al., 2025). KBA and IPA status therefore appears to be one comparatively modest driver of designation.

4.5. Implications for 30-by-30 and global conservation targets

KBAs and IPAs can make important contributions to the ecologically representative expansion of PAs. For the many countries that already incorporate such sites of high biodiversity value into their National Biodiversity Strategies and Action Plans, our findings provide a quantitative basis for combining these networks deliberately to close specific representation gaps. This complementarity may be most evident where existing protection is sparsest. The additional representation KBAs and IPAs offer tended to be greater in countries with limited PA coverage, so countries requiring the largest expansion to meet 30-by-30 may also be those for which these networks offer the most representational return per unit of new protection.

Protected area designation will not be feasible or appropriate everywhere. Many sites of high biodiversity value overlap with landscapes that are used by and important to people, where statutory protection is difficult or undesirable to establish (Fajardo et al., 2026; Yang et al., 2020). Furthermore, planners with limited resources may prioritise other conservation features that KBAs or IPAs do not inherently target, such as sites that are important for local ecosystem services (Smith et al., 2019). KBA or IPA status therefore does not automatically make a site an immediate management or protection priority – that depends on local threats, costs, opportunities and resourcing (Maxwell et al., 2020). There are, however, alternative pathways to protection. For example, other effective area-based conservation measures (OECMs) offer an alternative for securing sites of high biodiversity value that are unlikely to receive formal protection (Jonas, Bingham, et al., 2024). Indeed, the criteria for IPA identification explicitly acknowledge the provisioning services provided by plants (Darbyshire et al., 2017). This reflects wider recognition that meeting area-based targets through protected area expansion alone is unrealistic (Butchart et al., 2015).

IPAs remain far more geographically restricted than KBAs, with only a fraction of countries having active IPA programmes, yet our results suggest that scaling up IPAs could greatly increase the representation of plants in conservation planning. Extending the IPA programme to unassessed countries, and completing the reassessment of legacy KBAs against the KBA Standard, would therefore directly improve the evidence base for representative expansion – not only towards 2030, but for the area-based targets likely to succeed it. Given the limited time to meet 30-by-30, and that most national PA networks are built iteratively and opportunistically (Baldi et al., 2017), we suggest that using existing sites identified as important for biodiversity to guide representative expansion is both scientifically defensible and temporally realistic.

References

Anderson, S. (2002). Identifying Important Plant Areas: A Site Selection Manual for Europe, and a basis for developing guidelines for other regions of the world. Plantlife International.

Bachman, S. P., Brown, M. J. M., Leão, T. C. C., Nic Lughadha, E., & Walker, B. E. (2024). Extinction risk predictions for the world's flowering plants to support their conservation. New Phytologist. https://doi.org/10.1111/nph.19592

Baldi, G., Texeira, M., Martin, O. A., Grau, H. R., & Jobbágy, E. G. (2017). Opportunities drive the global distribution of protected areas. PeerJ, 5, e2989. https://doi.org/10.7717/peerj.2989

BirdLife International. (2024a). Digital boundaries of Important Bird and Biodiversity Areas from the World Database of Key Biodiversity Areas. March 2024 version [Data set]. BirdLife International. http://datazone.birdlife.org/site/requestgis

BirdLife International. (2024b). World Database of Key Biodiversity Areas. Developed by the KBA Partnership: BirdLife International, International Union for the Conservation of Nature, American Bird Conservancy, Amphibian Survival Alliance, Conservation International, Critical Ecosystem Partnership Fund, Global Environment Facility, Re:wild, NatureServe, Rainforest Trust, Royal Society for the Protection of Birds, Wildlife Conservation Society and World Wildlife Fund. June 2024 version [Data set]. http://keybiodiversityareas.org/kba-data/request

Brown, M. J. M., Bachman, S. P., & Nic Lughadha, E. (2023). Three in four undescribed plant species are threatened with extinction. New Phytologist, 240(4), Article 4. https://doi.org/10.1111/nph.19214

Butchart, S. H. M., Clarke, M., Smith, R. J., Sykes, R. E., Scharlemann, J. P. W., Harfoot, M., Buchanan, G. M., Angulo, A., Balmford, A., Bertzky, B., Brooks, T. M., Carpenter, K. E., Comeros-Raynal, M. T., Cornell, J., Ficetola, G. F., Fishpool, L. D. C., Fuller, R. A., Geldmann, J., Harwell, H., … Burgess, N. D. (2015). Shortfalls and Solutions for Meeting National and Global Conservation Area Targets. Conservation Letters, 8(5), Article 5. https://doi.org/10.1111/conl.12158

CBD. (2022). Decision Adopted by The Conference of The Parties to The Convention on Biological Diversity. In 15/4 Kunming-Montreal Global Biodiversity Framework. Convention on Biological Diversity. Secretariat of the Convention on Biological Diversity. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04- en.pdf

CBD. (2024). Guidance on using the indicators of the monitoring framework of the Kunming-Montreal Global Biodiversity Framework. In Subsidiary Body on Scientific, Technical and Technological Advice. Convention on Biological Diversity. Secretariat of the Convention on Biological Diversity. https://www.cbd.int/doc/c/92cf/b458/18519b4c0b487bf9bfc23988/sbstta-26-inf-14-en.pdf

Darbyshire, I., Anderson, S., Asatryan, A., Byfield, A., Cheek, M., Clubbe, C., Ghrabi, Z., Harris, T., Heatubun, C. D., Kalema, J., Magassouba, S., McCarthy, B., Milliken, W., De Montmollin, B., Lughadha, E. N., Onana, J.-M., Saïdou, D., Sârbu, A., Shrestha, K., & Radford, E. A. (2017). Important Plant Areas: Revised selection criteria for a global approach to plant conservation. Biodiversity and Conservation, 26(8), Article 8. https://doi.org/10.1007/s10531-017-1336-6

Delso, Á., Fajardo, J., & Muñoz, J. (2021). Protected area networks do not represent unseen biodiversity. Scientific Reports, 11(1), Article 1. https://doi.org/10.1038/s41598-021-91651-z

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

Duong, T. (2007). **ks**: Kernel Density Estimation and Kernel Discriminant Analysis for Multivariate Data in R. Journal of Statistical Software, 21(7). https://doi.org/10.18637/jss.v021.i07

Earth Resources Observation And Science (EROS) Center. (2017). Global 30 Arc-Second Elevation (GTOPO30) [Tiff]. U.S. Geological Survey. https://doi.org/10.5066/F7DF6PQS

Eken, G., Bennun, L., Brooks, T. M., Darwall, W., Fishpool, L. D. C., Foster, M., Knox, D., Langhammer, P., Matiku, P., Radford, E., Salaman, P., Sechrest, W., Smith, M. L., Spector, S., & Tordoff, A. (2004). Key Biodiversity Areas as Site Conservation Targets. BioScience, 54(12), Article 12. https://doi.org/10.1641/0006-3568(2004)054%5B1110:KBAASC%5D2.0.CO;2

Fajardo, J., Bingham, H. C., Brockington, D., Chaplin-Kramer, R., Fitzsimons, J. A., Fleischman, F., Frechette, A., Garrett, R. D., Hazin, C., Kuemmerle, T., Lessmann, J., Nuvunga, M. O. F., O'Donnell, B., Onyai, F., Pinto, R., Pfeifer, M., Pritchard, R., Ryan, C. M., Shyamsundar, P., … Sandbrook, C. (2026). Social implications of the 30×30 global conservation target. Nature Communications, 17(1), 4067. https://doi.org/10.1038/s41467-026-71860-8

Fick, S. E., & Hijmans, R. J. (2017). WorldClim 2: New 1-km spatial resolution climate surfaces for global land areas. International Journal of Climatology, 37(12), 4302–4315. https://doi.org/10.1002/joc.5086

Graham, C. H., Araujo, M. L., Barreto, E., Dambros, C. S., Diniz-Filho, J. A. F., Zimmermann, N. E., Rangel, T. F., & Coelho, M. T. P. (2025). Biodiversity Patterns Redefined in Environmental Space. Ecology Letters, 28(2), e70008. https://doi.org/10.1111/ele.70008

Habel, K., Grasman, R., Gramacy, R. B., Mozharovskyi, P., & Sterratt, D. C. (2025). geometry: Mesh Generation and Surface Tessellation (Version 0.5.2, p. 0.5.2) [Data set]. https://doi.org/10.32614/CRAN.package.geometry

Hijmans, Robert. J. (2023). terra: Spatial Data Analysis (Version R package version 1.7-55) [Computer software]. https://CRAN.R-project.org/package=terra

Hyndman, R. J. (1996). Computing and Graphing Highest Density Regions. The American Statistician, 50(2), 120–126. https://doi.org/10.1080/00031305.1996.10474359

Irl, S. D. H., Harter, D. E. V., Steinbauer, M. J., Gallego Puyol, D., Fernández-Palacios, J. M., Jentsch, A., & Beierkuhnlein, C. (2015). Climate vs. Topography – spatial patterns of plant species diversity and endemism on a high-elevation island. Journal of Ecology, 103(6), 1621–1633. https://doi.org/10.1111/1365-2745.12463

IUCN. (2016). A Global Standard for the Identification of Key Biodiversity Areas, Version 1.0 (First Edition). IUCN. https://portals.iucn.org/library/node/46259

IUCN. (2024). The IUCN Red List of Threatened Species. Version 2024-2. https://www.iucnredlist.org

Jago, S., Gizaw, G., Genanaw, B., Langley, J., Lulekal, E., White, J. D. M., Rowlands, A. N., Geda, T., Wakjira, K., Regassa, F., Demissew, S., Woldeyes, F., Abebe, W., Jones, J. P. G., Smith, R. J., & Borrell, J. S. (2026). Trade-offs between nature and people in Ethiopia's protected areas demonstrate challenges in translating global conservation targets into national realities. Nature Ecology & Evolution. https://doi.org/10.1038/s41559-026-03047-9

Jarvis, A., Reuter, H. I., Nelson, A., & Guevara, E. (2008). Hole-filled SRTM or the globe. Version 4, available from the CGIAR-CSI SRTM 90m Database [Data set]. http://srtm.csi.cgiar.org/

Jonas, H. D., Bingham, H. C., Bennett, N. J., Woodley, S., Zlatanova, R., Howland, E., Belle, E., Upton, J., Gottlieb, B., Kamath, V., Lessmann, J., Delli, G., Dubois, G., Ahmadia, G., Claudet, J., Cook, C., Deza, J., Grorud-Colvert, K., Gurney, G., … Ruiz, L. (2024). Global status and emerging contribution of other effective area-based conservation measures (OECMs) towards the '30x30' biodiversity Target 3. Frontiers in Conservation Science, 5, 1447434. https://doi.org/10.3389/fcosc.2024.1447434

Jonas, H. D., Wood, P., Woodley, S., & Volume Editors (Eds). (2024). Guidance on other effective area-based conservation measures (OECMs). IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/LAAW4624

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

KBA Standards and Appeals Committee of IUCN SSC and IUCN WCPA. (2022). Guidelines for using A global standard for the identification of Key Biodiversity Areas: Version 1.2 (1.2). IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2022.KBA.1.2.en

Kor, L., & Diazgranados, M. (2023). Identifying important plant areas for useful plant species in Colombia. Biological Conservation, 284, 110187. https://doi.org/10.1016/j.biocon.2023.110187

Kor, L., Perez, F., Inwood, K., Darbyshire, I., & Diazgranados, M. (2025). An evaluation of important plant areas around the world. Conservation Biology, e70013. https://doi.org/10.1111/cobi.70013

Kullberg, P., Di Minin, E., & Moilanen, A. (2019). Using key biodiversity areas to guide effective expansion of the global protected area network. Global Ecology and Conservation, 20, e00768. https://doi.org/10.1016/j.gecco.2019.e00768

Lansley, T. P., Crowe, O., Butchart, S. H. M., Edwards, D. P., & Thomas, G. H. (2025). Effectiveness of Key Biodiversity Areas in representing global avian diversity. Conservation Biology, e70000. https://doi.org/10.1111/cobi.70000

Lenth, R. V. (2016). Least-Squares Means: The R Package **lsmeans**. Journal of Statistical Software, 69(1). https://doi.org/10.18637/jss.v069.i01

Lewis, E., MacSharry, B., Juffe-Bignoli, D., Harris, N., Burrows, G., Kingston, N., & Burgess, N. D. (2019). Dynamics in the global protected-area estate since 2004. Conservation Biology, 33(3), Article 3. https://doi.org/10.1111/cobi.13056

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), Article 7828. https://doi.org/10.1038/s41586-020-2773-z

Moat, J. F., Bachman, S. P., & Walker, B. E. (2020). rCAT: Conservation Assessment Tools (Version 0.2.0) [Computer software].

Naidoo, R., Gerkey, D., Hole, D., Pfaff, A., Ellis, A. M., Golden, C. D., Herrera, D., Johnson, K., Mulligan, M., Ricketts, T. H., & Fisher, B. (2019). Evaluating the impacts of protected areas on human well-being across the developing world. Science Advances, 5(4), Article 4. https://doi.org/10.1126/sciadv.aav3006

Noroozi, J., Talebi, A., Doostmohammadi, M., Rumpf, S. B., Linder, H. P., & Schneeweiss, G. M. (2018). Hotspots within a global biodiversity hotspot—Areas of endemism are associated with high mountain ranges. Scientific Reports, 8(1), 10345. https://doi.org/10.1038/s41598-018-28504-9

Padgham, M., Lovelace, R., Salmon, M., & Rudis, B. (2017). Osmdata. The Journal of Open Source Software, 2(14), 305. https://doi.org/10.21105/joss.00305

Pironon, S., Etherington, T. R., Borrell, J. S., Kühn, N., Macias-Fauria, M., Ondo, I., Tovar, C., Wilkin, P., & Willis, K. J. (2019). Potential adaptive strategies for 29 sub-Saharan crops under future climate change. Nature Climate Change, 9(10), Article 10. https://doi.org/10.1038/s41558-019-0585-7

Pironon, S., Ondo, I., Diazgranados, M., Allkin, R., Baquero, A. C., Cámara-Leret, R., Canteiro, C., Dennehy-Carr, Z., Govaerts, R., Hargreaves, S., Hudson, A. J., Lemmens, R., Milliken, W., Nesbitt, M., Patmore, K., Schmelzer, G., Turner, R. M., Van Andel, T. R., Ulian, T., … Willis, K. J. (2024). The global distribution of plants used by humans. Science, 383(6680), 293–297. https://doi.org/10.1126/science.adg8028

Plantlife International. (2023). Important Plant Areas. https://www.plantlife.org.uk/protecting-plants-fungi/important-plant-areas/

Plumptre, A. J., Baisero, D., Brooks, T. M., Buchanan, G., Butchart, S. H. M., Bowser, A., Boyd, C., Carneiro, A. P. B., Davies, T., Elliot, W., Foster, M., Langhammer, P. F., Marnewick, D., Matiku, P., McCreless, E., Raudsepp-Hearne, C., Tordoff, A. W., Azpiroz, A. B., Trisurat, Y., & Upgren, A. (2024). Targeting site conservation to increase the effectiveness of new global biodiversity targets. One Earth, 7(1), 11–17. https://doi.org/10.1016/j.oneear.2023.12.007

Plumptre, A. J., Hayes, J., Baisero, D., Rose, R., Holness, S., Von Staden, L., & Smith, R. J. (2024). Strengths and complementarity of systematic conservation planning and Key Biodiversity Area approaches for spatial planning. Conservation Biology, 39(2), e14400. https://doi.org/10.1111/cobi.14400

Plumptre, A. J., Waliczky, Z., Baisero, D., Crowe, O., Kivono, J., Tobar, C., Toscano, M. G., Boulad, N., Costa, H., Davila, C., Dirou, S., Duarte, E., Fierro, K., Castellanos-Castro, C., Haddad, H., Holness, S., Maisels, F., Marnewick, D., Mbende, M., … Woodley, S. (2025). Changes in Key Biodiversity Area networks following national comprehensive assessments. Conservation Biology, e70151. https://doi.org/10.1111/cobi.70151

Poggio, L., De Sousa, L. M., Batjes, N. H., Heuvelink, G. B. M., Kempen, B., Ribeiro, E., & Rossiter, D. (2021). SoilGrids 2.0: Producing soil information for the globe with quantified spatial uncertainty. SOIL, 7(1), Article 1. https://doi.org/10.5194/soil-7-217-2021

Richards, S. L., Farooq, H., Matimele, H., Alves, T., Datizua, C., Langa, C., Massingue, A., Osborne, J., Rokni, S., De Sousa, C., & Darbyshire, I. (2023). Identifying Mozambique's most critical areas for plant conservation: An evaluation of protected areas and Important Plant Areas. Biotropica, 55(6), Article 6. https://doi.org/10.1111/btp.13265

Royal Botanic Gardens Kew. (2024). Tropical Important Plant Areas Explorer [Data set]. https://tipas.kew.org/

Smith, R. J., Bennun, L., Brooks, T. M., Butchart, S. H. M., Cuttelod, A., Di Marco, M., Ferrier, S., Fishpool, L. D. C., Joppa, L., Juffe-Bignoli, D., Knight, A. T., Lamoreux, J. F., Langhammer, P., Possingham, H. P., Rondinini, C., Visconti, P., Watson, J. E. M., Woodley, S., Boitani, L., … Scaramuzza, C. A. de M. (2019). Synergies between the key biodiversity area and systematic conservation planning approaches. Conservation Letters, 12(1), e12625. https://doi.org/10.1111/conl.12625

UNEP-WCMC & IUCN. (2024a). Protected Planet Report 2024. UNEP-WCMC and IUCN.

UNEP-WCMC & IUCN. (2024b). Protected Planet: The World Database on Protected Areas (WDPA) and Other Effective-Area Based Conservation Measures (WDOECM), August 2024. UNEP-WCMC and IUCN. https://www.protectedplanet.net

United Nations. (2015). Transforming Our World: The 2030 Agenda for Sustainable Development. United Nations. https://sdgs.un.org/goals

Venter, O., Magrach, A., Outram, N., Klein, C. J., Possingham, H. P., Di Marco, M., & Watson, J. E. M. (2018). Bias in protected-area location and its effects on long-term aspirations of biodiversity conventions. Conservation Biology: The Journal of the Society for Conservation Biology, 32(1), Article 1. https://doi.org/10.1111/cobi.12970

Visconti, P., Butchart, S. H. M., Brooks, T. M., Langhammer, P. F., Marnewick, D., Vergara, S., Yanosky, A., & Watson, J. E. M. (2019). Protected area targets post-2020. Science, 364(6437), Article 6437. https://doi.org/10.1126/science.aav6886

Yang, R., Cao, Y., Hou, S., Peng, Q., Wang, X., Wang, F., Tseng, T.-H., Yu, L., Carver, S., Convery, I., Zhao, Z., Shen, X., Li, S., Zheng, Y., Liu, H., Gong, P., & Ma, K. (2020). Cost-effective priorities for the expansion of global terrestrial protected areas: Setting post-2020 global and national targets. Science Advances, 6(37), Article 37. https://doi.org/10.1126/sciadv.abc3436

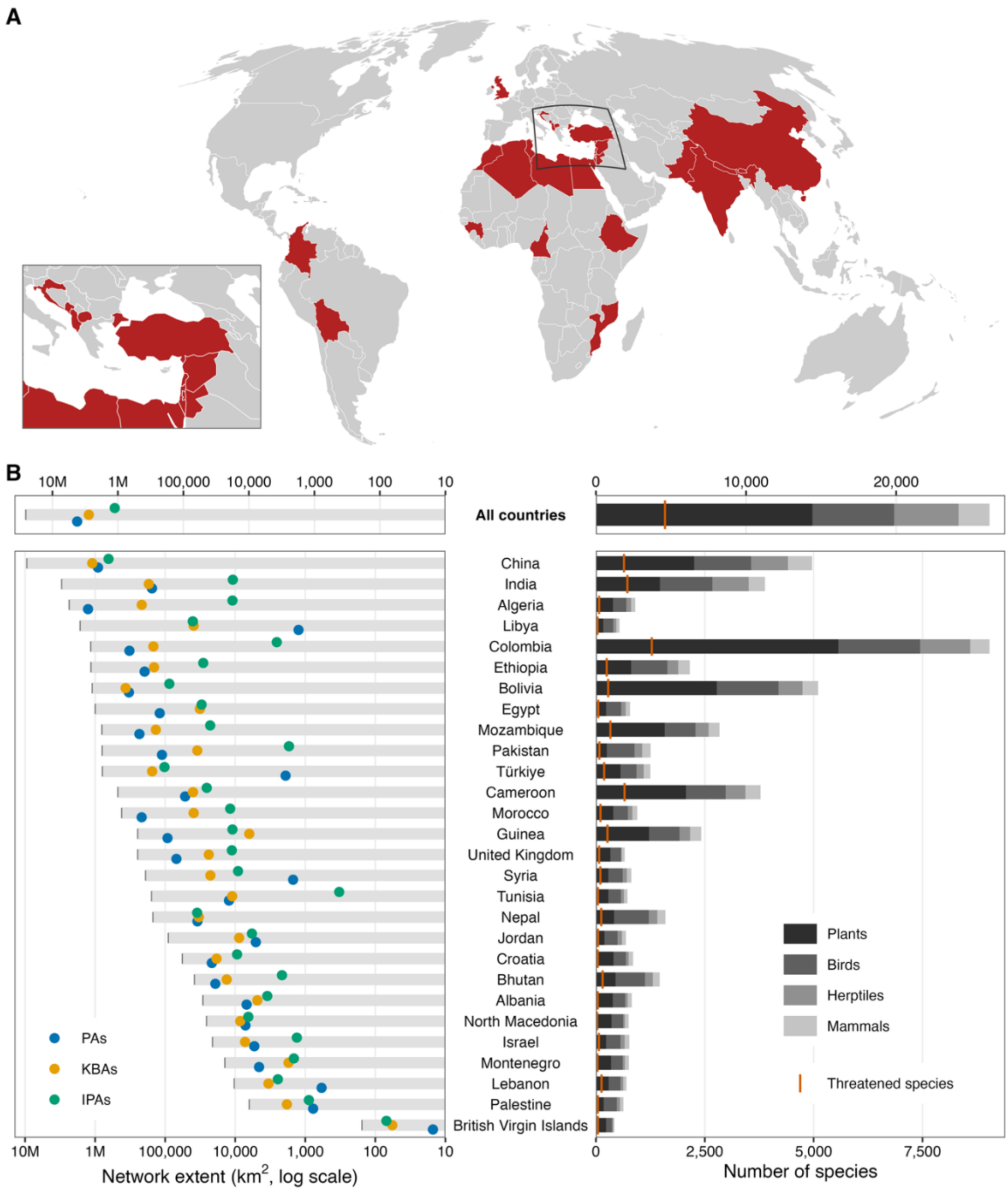

Figure 1. (A) The 28 studied countries (red) in Mollweide equal-area projection; the inset enlarges the eastern Mediterranean and Balkans, where several small, studied countries are located. **(B)** For each country: (left) the geographic extent of protected and conserved area (PA), key biodiversity area (KBA) and important plant area (IPA) networks (coloured points) against total national land area (grey bars); and (right) the number of plant, bird, herptile (reptile and amphibian) and mammal species with ranges intersecting the country (stacked bars), with threatened species (CR, EN, VU) marked in orange. Countries displayed in descending order of total land area.

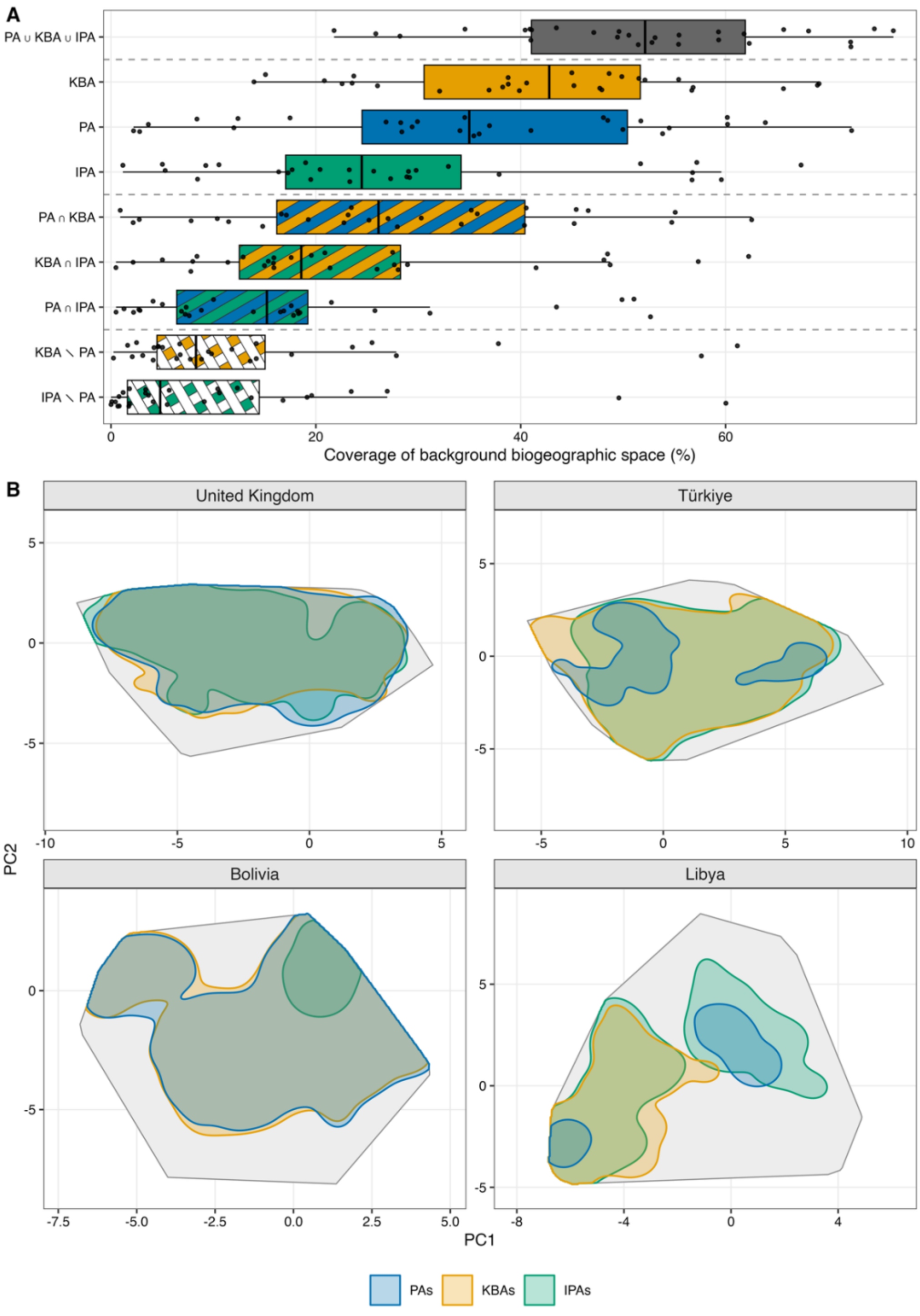

Figure 2. Representation of biogeographic space by protected and conserved area (PA), key biodiversity area (KBA) and important plant area (IPA) networks. Each country's biogeographic space is defined by the first three principal components of 13 environmental variables. Coverage is the volume of a network's 99% highest density region (HDR) of a kernel density estimate, expressed as a percentage of the volume of the convex hull of that country's background points. **(A)** Coverage across the 28 studied countries for each network individually, pairwise intersections (∩) of each network, the union (∪) of all three, and additionality (∖) by KBAs and IPAs. Boxes show the median and interquartile range, and points are individual countries. **(B)** Two-dimensional 99% HDRs for four countries showing contrasting biogeographic space coverage patterns, with each network's coverage shown above the panel. The grey hull and dashed line mark the background convex hull, against which network coverage is measured.

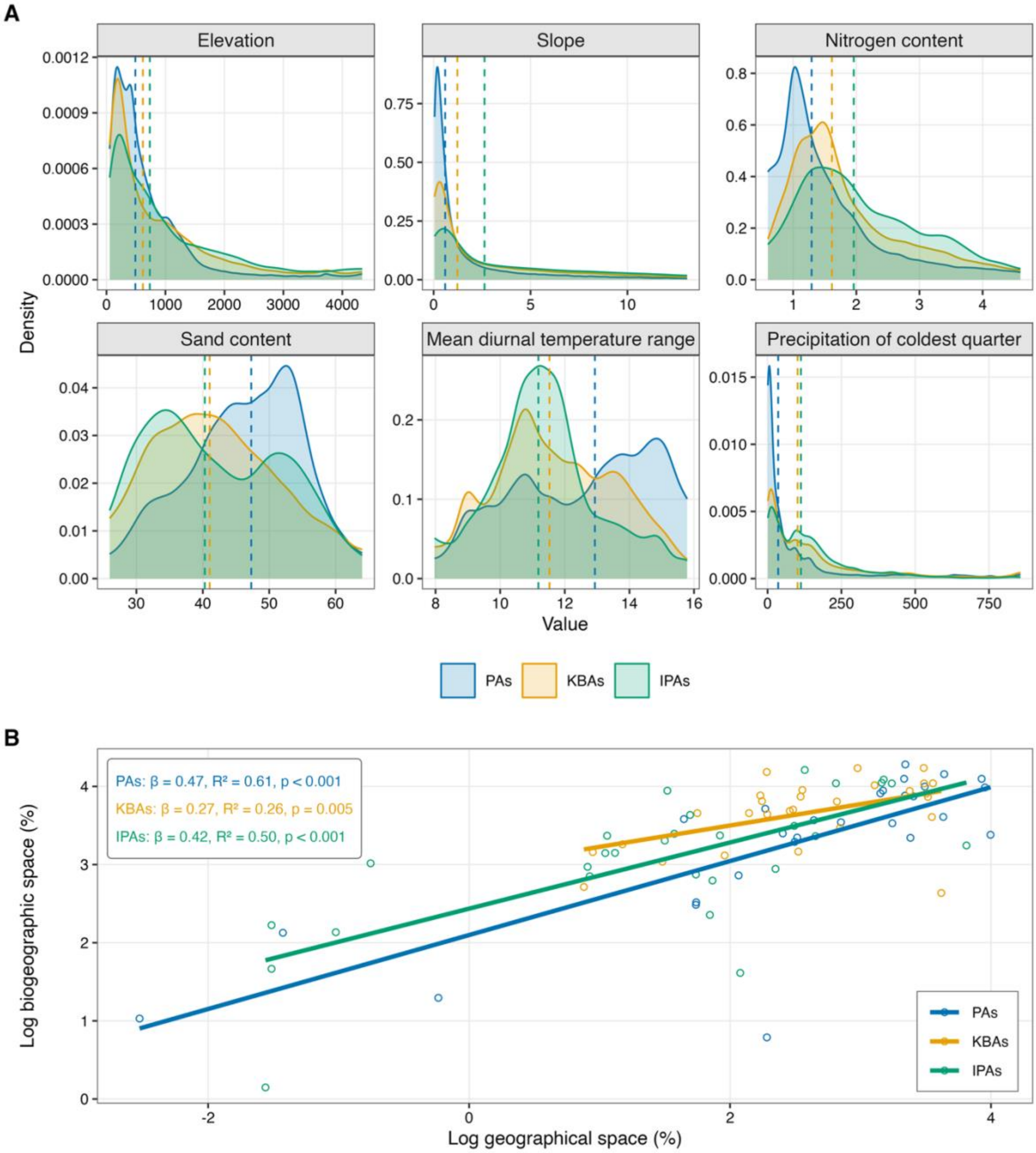

Figure 3. (A) Density distribution of six of the 13 retained environmental variables across protected and conserved areas (PAs), key biodiversity areas (KBAs) and important plant areas (IPAs), pooled across the 28 studied countries. Dashed vertical lines indicate each network's median value. Each panel's x-axis is truncated to the variable's 5^{th}-95^{th} percentile for visibility. **(B)** Relationship between log-transformed geographic and environmental coverage for each network across the 28 studied countries (one point per country per network). Environmental coverage is the volume of a network's 99% highest density region in biogeographic space of the first three principal components, expressed as a percentage of the volume of that country's background convex hull.

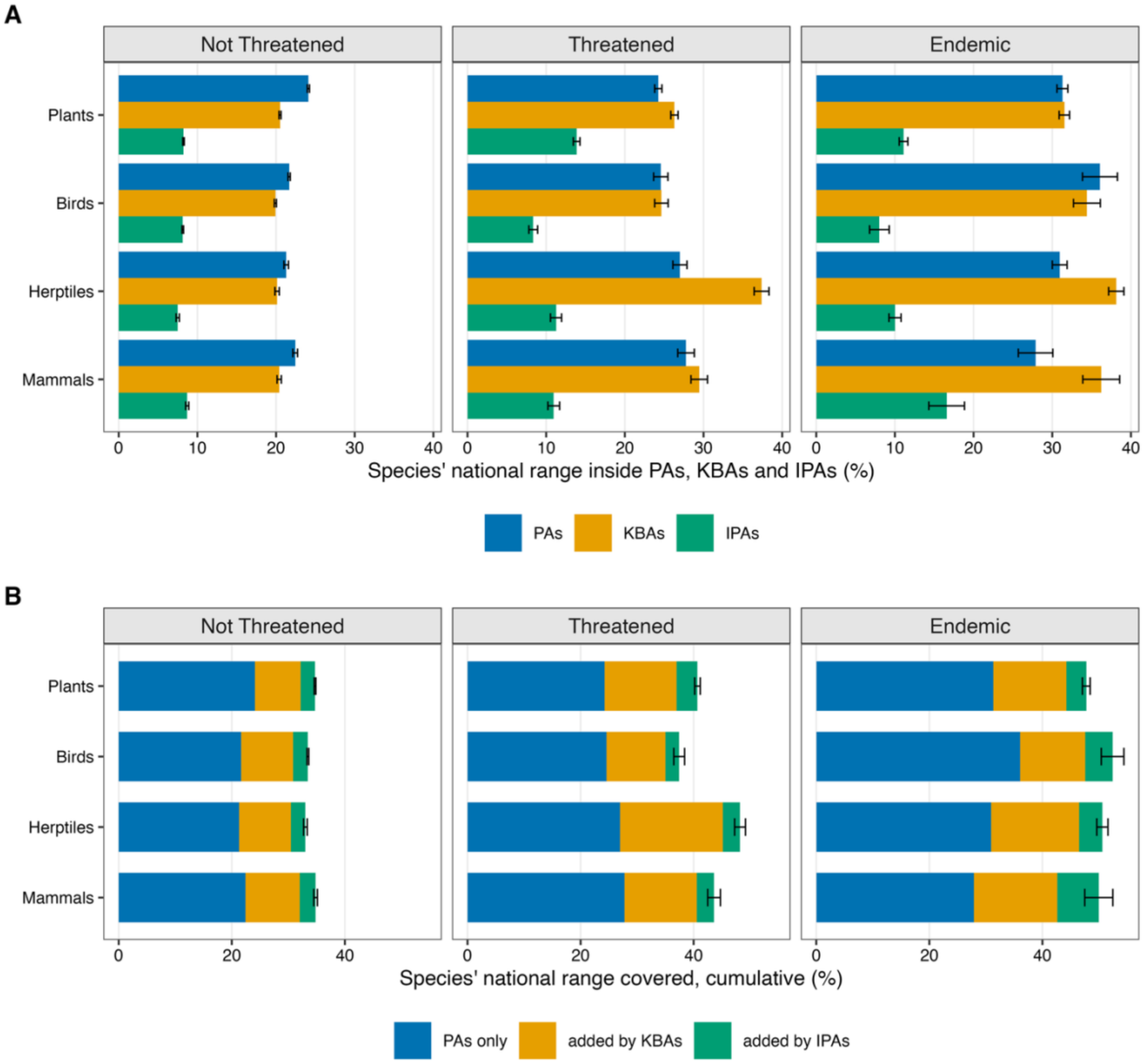

Figure 4. Representation of threatened (CR, EN, VU), not threatened (NT, LC) and endemic plant, bird, herptile (reptile and amphibian) and mammal species' ranges by protected and conserved areas (PAs), key biodiversity areas (KBAs) and important plant areas (IPAs) across the 28 studied countries. **(A)** Mean percentage of a species' national range falling inside each network, by taxon and species group. Error bars indicate ±1 standard error. **(B)** Cumulative species range coverage by the three networks combined. Bars show coverage by PAs, then the additional coverage contributed by KBAs, and finally any further coverage provided by IPAs, so total bar length is the mean percentage of range covered by the union of all three networks.

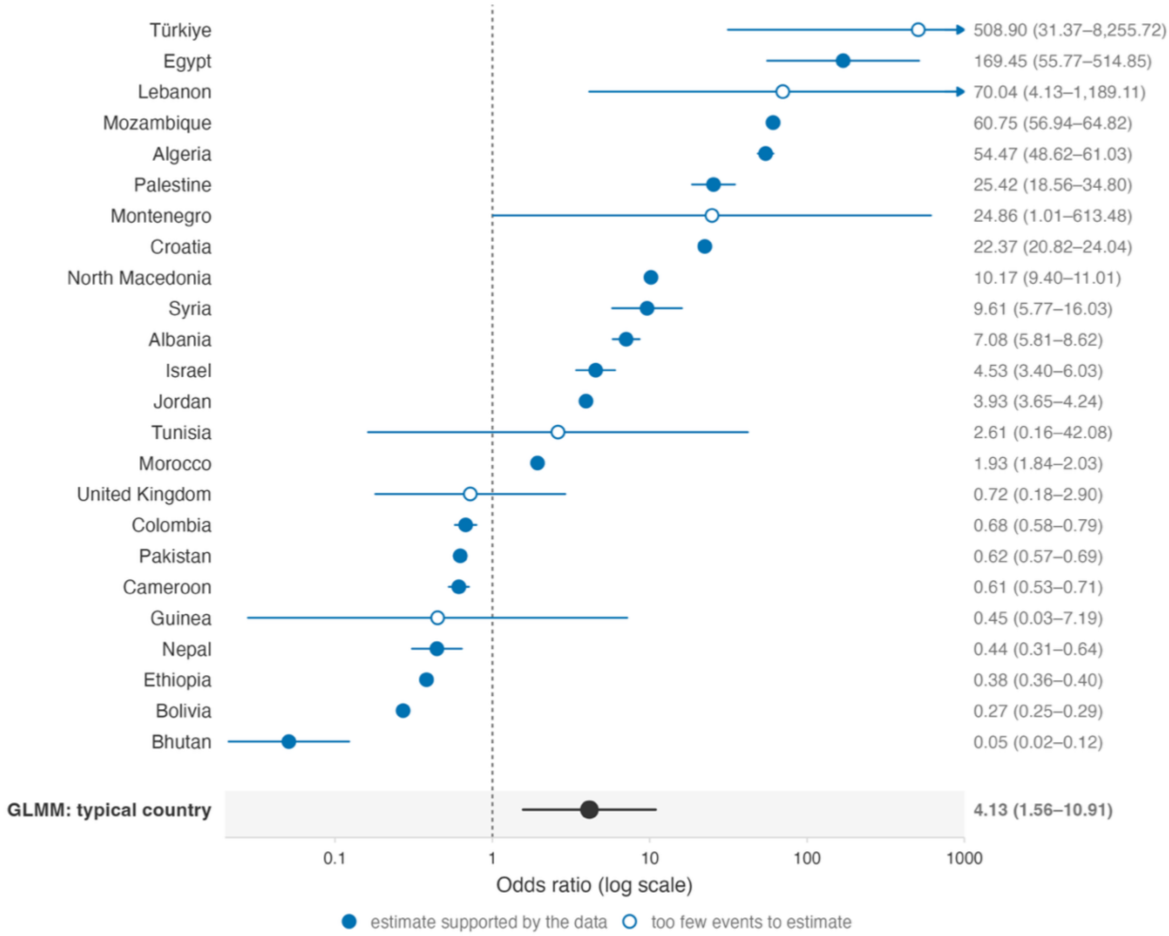

Figure 5. Odds that land identified as a key biodiversity area (KBA) or important plant area (IPA) by 2010 became subsequently designated as a protected and conserved area (PA) by 2024, relative to land outside these two networks in the same country. Each row is a country, displayed in descending order of odds ratios, derived from a 2 x 2 contingency table of 1 km grid cells. Points are odds ratios and horizontal lines represent 95% confidence intervals. Points to the right of the dashed vertical line (odds ratio = 1) indicate that cells inside KBAs/IPAs were more likely to become PAs than other land. Open points indicate estimates with too few designation events to interpret individually, while arrowheads mark upper confidence intervals that exceed the x-axis scale. The shaded row is the summary estimate from a binomial generalised linear mixed model, in which KBA/IPA effect was allowed to vary among countries.

Table 1. Coverage of biogeographic space by protected and conserved areas (PAs), key biodiversity areas (KBAs) and important plant areas (IPAs). Biogeographic space was defined by a principal component analysis of 13 environmental variables, separately for each studied country. Coverage is the volume of a network's 99% highest density region, as a percentage of the volume of the convex hull of that country's background points. Countries are displayed in descending order of their combined coverage by all three networks. Mean and SD across all 28 studied countries are reported below.

	Coverage by individual networks (%)			**Coverage shared between pairs of networks (%)**			**All three networks combined (%)**
Country	PA	KBA	IPA	PA ∩ KBA	PA ∩ IPA	KBA ∩ IPA	PA ∪ KBA ∪ IPA
United Kingdom	72.3	65.7	51.7	62.6	49.9	48.5	76.4
Türkiye	8.4	69.0	67.4	7.8	7.3	62.3	74.5
North Macedonia	60.2	49.9	56.7	40.4	51.1	41.5	72.2
Lebanon	12.4	69.1	59.6	11.5	10.0	57.3	72.2
Croatia	63.9	56.7	56.8	55.1	52.7	48.7	68.1
Nepal	50.0	55.4	57.2	46.6	43.5	48.1	65.7
Bhutan	60.2	56.8	17.7	54.8	17.6	17.6	62.3
Albania	51.7	45.0	32.9	35.2	31.1	27.7	61.8
Morocco	53.9	45.2	23.3	40.5	21.5	21.4	59.3
Algeria	29.4	48.6	20.4	23.2	1.3	15.9	59.3
Colombia	48.5	52.1	9.3	45.4	8.8	8.4	55.5
Montenegro	54.5	23.7	19.0	23.5	18.2	15.9	55.4
Ethiopia	34.5	47.9	17.3	30.3	13.9	15.9	53.1
Palestine, State of	28.4	51.5	28.9	28.0	16.5	28.0	52.8
Cameroon	29.9	38.8	37.9	25.2	18.3	29.0	51.5
Bolivia †	48.1	47.7	5.0	45.2	5.0	5.0	50.6
Guinea	37.0	23.5	27.3	17.1	16.9	15.4	49.6
Israel	34.1	36.9	10.5	22.7	6.9	8.0	49.5
China	41.0	39.9	29.2	35.8	25.8	26.0	47.1
Syrian Arab Republic	3.7	40.6	29.8	2.8	2.8	27.5	43.5
Jordan	12.0	38.2	16.4	10.4	7.3	15.0	41.1
Mozambique	28.3	32.1	29.0	27.1	18.5	20.9	41.0
India	35.9	38.8	8.5	34.2	7.6	7.8	40.9
Pakistan	35.4	20.8	5.3	16.6	4.1	2.2	40.4
Egypt	26.9	26.0	23.3	19.3	17.8	19.6	34.6
Virgin Islands, British	2.2	14.0	25.7	2.2	2.2	11.4	28.2
Tunisia	17.5	22.6	1.2	14.8	0.5	0.5	25.9
Libya	2.8	15.1	19.5	0.9	2.7	12.9	21.8
Mean	35.1	41.8	28.1	27.8	17.1	23.5	51.9
SD	19.7	15.5	18.4	16.8	15.5	16.8	14.6

† Plurinational State of Bolivia

Table 2. Mean proportion of species' national ranges intersecting protected and conserved areas (PAs), key biodiversity areas (KBAs) and important plant areas (IPAs) in each of the 28 studied countries. For each species in each country, the proportion of its national range falling inside a network was calculated, and values averaged across all species in that country. Countries are displayed in descending order of the mean proportion covered by all three networks. Mean and SD across all 28 studied countries are reported below.

	Coverage by individual networks (%)			**Coverage by pairs of networks combined (%)**			**All three networks combined (%)**
Country	PA	KBA	IPA	PA ∪ KBA	PA ∪ IPA	KBA ∪ IPA	PA ∪ KBA ∪ IPA
Bhutan	43.0	35.2	8.4	51.8	45.7	38.0	54.0
Morocco	47.9	14.2	4.0	52.7	49.6	15.8	53.5
North Macedonia	32.2	35.0	26.7	48.1	39.9	44.2	52.4
Bolivia †	34.8	42.8	8.1	50.2	36.8	44.8	51.1
Israel	21.6	31.3	11.3	45.6	28.7	37.3	49.6
Virgin Islands, British	8.2	33.8	41.2	34.6	41.6	48.0	48.5
Croatia	41.0	36.0	18.5	43.7	44.0	38.6	45.0
Algeria	34.4	18.9	8.7	44.3	40.2	19.0	44.4
Nepal	21.1	24.4	21.8	28.7	36.0	38.8	41.7
Albania	29.1	22.2	15.5	36.7	34.2	27.4	38.7
Mozambique	29.7	23.0	9.2	36.0	34.8	26.0	38.5
Montenegro	33.9	16.4	13.0	37.2	35.1	18.7	37.8
Colombia	31.4	18.9	0.5	37.0	31.7	19.4	37.3
United Kingdom	33.8	10.1	7.1	34.7	34.2	13.9	35.0
Lebanon	6.0	32.5	23.7	34.1	27.1	33.4	35.0
Ethiopia	21.4	16.8	2.4	32.5	23.5	18.0	33.7
Türkiye	0.53	27.4	21.0	27.4	21.2	33.1	33.1
Jordan	7.1	26.3	11.9	28.5	17.3	30.7	32.8
Palestine, State of	10.9	26.1	12.3	27.7	19.6	30.8	31.7
Guinea	29.5	7.3	7.02	30.1	31.0	10.1	31.5
Syrian Arab Republic	2.30	26.1	15.2	27.0	16.9	26.9	27.8
China	8.2	12.6	13.4	16.4	19.1	22.5	25.4
Egypt	14.9	14.2	9.3	23.6	18.9	16.8	25.0
Libya	1.03	20.4	12.4	20.8	13.2	21.9	22.3
Cameroon	14.2	12.0	11.7	18.8	20.2	16.0	22.2
India	13.6	18.4	1.4	21.6	14.4	19.3	22.2
Pakistan	15.0	7.1	1.1	18.6	15.9	8.1	19.4
Tunisia	6.9	9.8	1.1	11.7	7.3	10.0	11.9
Mean	21.2	22.1	12.1	32.9	28.5	26.0	35.8
SD	13.8	9.7	9.0	11.2	11.3	11.3	11.3

† Plurinational State of Bolivia