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# Roadless areas and the 30% protection target: contrasting national prospects in the Iberian Peninsula

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## Abstract

The European Union has committed to protecting 30% of its land area by 2030, and Member States must now submit national restoration plans under Regulation (EU) 2024/1991. Meeting these commitments requires identifying land that is already ecologically intact but lies outside formal protection. Roadless areas (RLAs) are the largest such reserve in Europe, yet remain largely absent from planning instruments. We assess how far RLAs could close the protection gap in Spain and Portugal, quantifying their contribution to the existing network of protected areas (PAs), comparing their environmental characteristics using generalised linear models, and ranking individual roadless patches by their potential contribution. Both countries fall below the 30% objective, with PAs covering 27.9% of Spain and 22.3% of Portugal; incorporating RLAs would raise coverage to 44.9% and 34.3% respectively. Protection is strongly biased uphill, reaching 15.7% of land below 200 m but 87.0% above 2000 m, and the models explain why; PAs were best predicted by terrain, RLAs by landscape structure and biodiversity. The largest gain falls on agro-forestry systems, the dehesas and montados of the south-west Iberia, which rise by more than 30 percentage points in both countries. Prioritising 13229 km² of roadless land, 2.7% of Spanish territory, would suffice for Spain to meet the objective, whereas Portugal cannot reach it through roadlessness alone. RLAs therefore offer Member States an already-intact land reserve for meeting EU biodiversity targets, and we identify where it is most valuable.

## 1. Introduction

Rapid technological development, population growth and rising per capita resource consumption have driven the loss of natural areas in favour of agricultural and urbanised land and of human-made infrastructure (Johnson et al. 2017; Van Dyke and Lamb 2020). Road networks in particular are becoming increasingly dense worldwide (Selva et al. 2011; Zomeni and Vogiatzakis 2014), with a projected growth of 3.0–4.7 million km by 2050 (Meijer et al. 2018). Their effects on biodiversity are well documented and diverse, ranging from habitat loss, fragmentation and vehicle collisions to noise, light and chemical pollution, altered animal behaviour, invasion by alien species and changes in hydrology and microclimate (Fahrig 2003; Crist et al. 2005; Laurance 2015; Selva et al. 2015; Helldin 2019). Together this drive landscape homogenisation, population declines and the alteration of ecosystem services (Forman and Alexander 1998; Benítez-López et al. 2010; Gardiner et al. 2018; Phillips et al. 2020), and both paved and unpaved roads can cause considerable damage (Sillero 2008; Shanley and Pyare 2011). Because impacts depend on road density, traffic intensity and landscape context, each situation is particular and universal mitigation measures are difficult to establish (Geneletti 2003; Freudenberger et al. 2013; Bennett 2017).

Road impacts on biodiversity can be especially critical in areas with high natural and cultural values, such as protected areas (PAs) (Garriga et al. 2012; Helldin 2019). Overall, PAs and protected landscapes are essential for conserving natural and cultural biodiversity, some of them reflecting sustainable land use practices (CBD 2022). Therefore, PAs are the main tool to conserve biodiversity and maintain ecosystems' integrity (Geldmann et al. 2013; Venter et al. 2014; Visconti et al. 2019; Pacifici et al. 2020). However, their ecological interest and proximity to human settlements expose them to pressures, including resource exploitation and ecotourism associated with heavy traffic (Garriga et al. 2012; Tverijonaite et al. 2018). Such traffic and the associated networks contribute to habitat fragmentation (Angulo et al. 2016), as roads both limit species movement and grant visitors access to undisturbed areas (Coffin 2007; Hadwen et al. 2008).

Beyond these pressures, the extent of the network of PAs is now subject to binding policy commitments. Target 3 of the Kunming-Montreal Global Biodiversity Framework and the EU Biodiversity Strategy for 2030 both commit signatories to protecting at least 30% of terrestrial area by 2030 (European Commission 2020; CBD 2022). These commitments are no longer aspirational: under Regulation (EU) 2024/1991 on nature restoration, Member States are required to submit national restoration plans setting out the area-based measures through which they will meet their obligations (European Union 2024). Closing the remaining gap by restoring degraded land is slow and costly, whereas land that is already ecologically intact but lacks formal designation could contribute immediately. Identifying where such land occurs, and how much of the gap it could close, is therefore a practical and time-sensitive need rather than a long-term objective.

Roadless areas (RLAs) are natural areas with no roads or with low-traffic intensity (below an established limit), and therefore experience relatively low levels of human impact (Selva et al. 2015; Psaralexi et al. 2017). RLAs can occur inside or outside protected areas, yet are often overlooked in conservation planning (Selva et al. 2015). However, they could play a key role, their environmental characteristics often resemble those of PAs, while also extending natural corridors (DeVelice and Martin 2001; Crist 2005; Raymond et al. 2026) or by hosting a high species richness (Rodrigues et al. 2004; Chen and Roberts 2008). In addition, they are a fundamental piece in the fight against climate change thanks to the carbon fixing processes and the cushioning effect they provide (Selva et al. 2011; Psaralexi et al. 2017; Fan et al. 2025). In Europe, RLAs cover more than a third of EU territory, many included within the Natura 2000 network (European Commission, 2020). However, PAs–RLA overlap differs strongly among member states. Considering that roads are major drivers of landscape fragmentation, reducing connectivity and increasing species' genetic isolation (Forman and Alexander 1998; Bennett 2017), the addition of RLAs into existing conservation networks can help maintain or enhance landscape connectivity (Crist et al. 2005; D'Amico et al. 2016) and protected-area integrity, among other benefits (Torres et al., 2016; Psaralexi et al., 2017). Testolin et al. (2026) recently mapped EU roadless areas from OpenStreetMap and assessed their contribution to the 10% strict-protection objective. Their continental, abiotically driven prioritisation is complementary to the national-scale, representation-based approach adopted here, which addresses the 30% coverage objective at the level at which Member States report.

From a landscape ecology perspective, the contribution of RLAs to networks of PAs depends strongly on landscape structure, particularly patch area, isolation, and aggregation (Table S1) (McGarigal et al. 2012). Larger habitat patches generally support higher species richness, larger populations, species with broader home ranges, and more intact ecological processes (Keller and Sullivan 2023). In addition, landscapes composed of multiple habitat patches may enhance population resilience and facilitate movement through stepping-stone dynamics. Building on these principles, and on the policy need to identify where additional protection would deliver most, we addressed three questions:

Q1 (Representation). Which gaps in the representation of habitat types and elevation bands does the current network of PAs leave open, and which of them would the incorporation of RLAs close? We assess this against the 30% coverage threshold adopted in the EU Biodiversity Strategy for 2030 and in Target 3 of the Kunming-Montreal Global Biodiversity Framework.

Q2 (Spatial configuration). How does the incorporation of RLAs reorganise the spatial structure of the conservation network, in terms of area, isolation and aggregation, and which RLAs contribute most per unit area?

Q3 (Environmental profile). Are RLAs environmentally equivalent to existing PAs, or are the two governed by different environmental drivers? A negative answer would establish RLAs as complementary rather than substitutable conservation targets.

Although the role of RLAs in enhancing the effectiveness of networks of PAs has been repeatedly advocated (Selva et al. 2011; Ibisch et al. 2016; Kati et al. 2020), their conservation status and ecological characteristics relative to PAs remain poorly quantified (but see Crist et al. 2005; Psaralexi et al. 2017; Talty et al. 2020), and no assessment has yet translated them into the coverage thresholds that Member States are now required to report against. We therefore evaluate whether, and where, RLAs can close the protection gap in the Iberian Peninsula (Spain and Portugal), a region that combines a global biodiversity hotspot with one of the densest road networks in Europe. Specifically, we aimed to: 1) quantify the contribution of RLAs to the spatial extent, habitat representation and landscape configuration of the network of PAs, and benchmark the result against the 30% coverage threshold for each country; 2) compare the environmental features of RLAs and existing PAs using generalised linear models, in order to establish whether the two are governed by the same environmental drivers; and 3) identify the RLAs whose incorporation would deliver the largest gains in representation and network structure.

## Study area

To perform our analysis, we selected the Iberian Peninsula (IP; southern Europe) for two main reasons: first, it is one of the world top-ranked hotspot of biodiversity (Myers et al. 2000); and second, both countries rank among those with the largest road networks. The IP (582860 $km^2$) includes Portugal (PT; 89015 $km^2$) and Spain (SP; 492175 $km^2$). It is characterised by strong geologic, topographic and climatic gradients, from the temperate Atlantic north-west to the dry warm Mediterranean south-east, supporting high biodiversity and endemicity (Underwood et al. 2009). PAs across the IP harbour about 50% of European plant and terrestrial vertebrate species and > 30% of European endemic species (Araújo et al. 2007). There is a total of 627 PAs in Portugal which represents the 22.3% of terrestrial area, while Spain hosts a total of 5551 PAs that reaches the 27.5% of terrestrial area (reported by BISE 2024); our own estimate on the 1-km grid is 27.9%). Both countries also rank among the sixty with the longest road networks worldwide (BISE 2024).

## 2. Data and Methods

**Roadless and protected areas**

Following the scheme showed in Fig. 1a, we used the European section of the global raster map of RLAs (Ibisch et al. 2016), and their conservation status, for the ecological assessments. This dataset applies a 1 km buffer to all roads. It includes an assessment of their RLA status, quality, and coverage extent by PAs. We cropped then the RLA data for the IP with the $crop´functionofthe$raster´ R package (Hijmans et al. 2011). We also obtained a raster map of the PAs for the IP from the European inventory of Nationally Designated Areas (NatDA) and Natura 2000 network (McSpadden 2014). Both datasets were available at 1 × 1 km resolution (Fig. 1b and Table 1). Grids were binarised, with protected and roadless cells coded as 1 and the rest as 0.

We merged both datasets (RLAs∪PAs) with the $mer \geq ´functionomthe$raster´ R package (Hijmans et al. 2011) (Fig. 1b). We intersected both RLA and the PAs datasets (RLA∩PAs) with the $overlay´function(Hijmansetal.\ 2011)(Tab \leq 1).\ Thesegrwereb \in arised, withπxelscodedas\frac{1}{0}.\ F \in ally, weused$crop´ to extract RLAs in isolation (RLAs−PAs) and PAs in isolation (PAs−RLAs) (Hijmans et al. 2011).

**Table 1.** Summary of the dependent variables, data sources and occupied surfaces in the Iberian Peninsula. Areas were computed on a 1-km equal-area grid (ETRS89-LAEA, EPSG:3035); total terrestrial surface = 581933 $km^2$.

| Dependent variable | Code | Source | Occupied surface ($km^2$) | % of the total surface |
|---|---|---|---|---|
| Roadless areas | RLAs | http://www.roadless.online/ | 146302 | 25.14 |
| Protected areas | PAs | https://www.eea.europa.eu/ | 157302 | 27.03 |
| Merge between PAs and RLAs | RLAs∪PAs | Own elaboration | 252060 | 43.31 |
| Overlap between PAs and RLAs | RLAs∩PAs | Own elaboration | 51544 | 8.86 |
| RLAs in isolation | RLAs-PAs | Own elaboration | 94758 | 16.28 |
| PAs in isolation | PAs-RLAs | Own elaboration | 105758 | 18.17 |

**Environmental predictors**

To characterise and compare the features from RLAs against the ones from PAs, we selected different environmental variables grouped by habitat type, topography, and biodiversity/landscape attributes (Table S2). The set of predictors used for GLM (see below) included:

i Habitat type (HAB). We selected the most representative habitats in the IP, in terms of vegetation structure and composition, from the Corine Land Cover (CLC) 2018 Version 2020_20u1 (3-level) (Land 1992), in raster format at 100 m of pixel size.

ii Topography (TOPO). We derived a set of topographic attributes from the European digital elevation model (DEM) (EU-DEM 2015), at 30 m resolution by using the R package `RSAGA´ v1.3.0 (Conrad et al. 2015).

iii Biodiversity (BIO). We derived several terrestrial biodiversity indicators (Species richness, Endemicity, and Shannon diversity index) from the global database The World Bank (Harm 2005) at 1 × 1 km of pixel size.

iv Landscape connectivity (LAND). We computed several landscape variables (Patch mean area, Related circumscribing circle, and Distance to the nearest neighbour) from the CLC 2018 Version 2020_20u1 (3-level) database with a spatial resolution of 100 m by using the R package `landscapemetrics´ (Hesselbarth et al. 2019) and based on a raster mask at 1 × 1 km of pixel size.

The fine-scale predictors were aggregated at 1 km by the mean. All predictors were projected to WGS84/UTM30 at 1 × 1 km to match the response variables. Analyses were performed in R v4.3.3 (R Core Team, 2021) and QGIS v3.18.3 (QGIS - Odense 2021) and ArcGIS v10.8.1 (ESRI 2021) to conduct the spatial analyses.

**Quantification of the extent contribution of roadless areas to protected areas**

To describe the influence of the RLAs in the preservation of natural areas, we measured the proportion of RLA extension on the existing PAs. For this, we determined the percentage of coincident pixels in the intersection map (RLAs∩PAs) between both RLAs and PAs and, therefore, the RLA contribution over the areas reserved for conservation. To further verify this contribution, we also analysed the coincident pixels among the different types of natural habitats (HAB) selected in previous steps, and of different elevation classes extracted from the elevation variable (TOPO). We reclassified elevation every 200 m, as in broader classes the variation was not appreciable accurately. We reclassified elevation with the $rey´function, whi \leq weextractedthecont \in \ \geq ncymatrixofoverlap\pi ngareaswiththe$crosstab´ function, both within the `raster´ package in R (Hijmans et al. 2011).

**Landscape connectivity assessment**

We selected three main elements of landscape connectivity to quantify the effect of the inclusion of RLAs into PAs: area, isolation, and aggregation (Table S1) (McGarigal et al. 2012). We computed six landscape metrics (Table S2) at the level of the grid cell for RLAs, PAs, both PAs and RLAs merged grid (RLAs∪PAs), the RLAs and PAs intersection grid (RLAs∩PAs), and the RLAs (RLAs-PAs) and PAs (PAs-RLAs), both in isolation, by using the R package $l \text{ and } s \cap emetrics´(Hesselb\bar{l}hetal.2019). To assessareaeffects, weusedthemetricsClass(Total)Area(CA), Percenta \geq ofL \text{ and } s \cap$ lsm´ R function returns the mean (MN) and standard deviation (SD). We then compared metric changes after adding RLAs to PAs across the three grids.

**Characterisation of protected and roadless areas**

We established three types of models representing the effects of the independent sets of predictors described above: i) Habitat (HAB); ii) Topography (TOPO); and iii) Biodiversity + Landscape connectivity (BIOLAND), for each of the six response variables (PAs, RLAs, RLAs∪PAs, RLAs∩PAs, RLAs-PAs, PAs-RLAs) (Table 1). We then built a fourth model type, iv) BEST, by combining the most contributing predictors selected from the previous three groups. In all model sets we used the same number of predictors to keep model complexity balanced. Model comparison was performed exclusively among predictor sets fitted to the same response variable. Because the six response variables are different binary surfaces, their likelihoods are not commensurable, and information criteria were therefore never compared across response variables. We also note that, with n ≈ 583000 grid cells, absolute differences in AIC are large by construction; we therefore interpret them jointly with the coefficient of discrimination (Tjur's D) rather than as decisive on their own.

*Model fitting*

We related the response variables (Table 1) to predictor variables (Table S2) using generalised linear models (GLMs), a flexible framework for ecological linear relationships (Guisan et al. 2002). We fitted a binomial logistic regression with the $glm´functionomtheR$stats´ package and logit link (Guisan et al. 2002), as the response variables (RLAs and PAs) are binary (presence/non-presence). Model performance was evaluated with the `performance´ R package (Lüdecke et al. 2021), which computes metrics of model quality (e.g., $R^2$, RMSE, overdispersion checks). We also examined the significance and contribution of each predictor per model.

We removed collinear predictors with Spearman's pair-wise correlation coefficients ≥ 0.7 or Variance Inflation Factors > 4 (Dormann et al. 2013; Becker et al. 2015), retaining six independent predictors per model type (Fig. S1 and S2).

*Predictor contributions on model sets and response curves*

To determine the most contributing predictors to the models of RLAs and PAs, and help us build the fourth model type (BEST), we calculated the importance of each predictor in each model type (Araújo and New 2007), ranging between 0 (no importance) and 1 (high importance), by using the $varImp´functionwith \in the$caret´ R package (Kuhn 2008). Only predictors with importance > 0.1 were retained. We then obtained response curves of the three most contributing predictors in the RLAsBEST and PAsBEST models with the $stat_sm\infty th´functionom$ggplot2´, and interpreted the maximum sensitivity range and the sign of the slope of the most important predictor. Differences between RLA and features of PAs across the most contributing predictors were summarised with t-tests; given the sample size and the spatial dependence between neighbouring cells, these are reported as descriptive summaries of effect size rather than as inferential tests.

**Representation gaps and spatial prioritisation of roadless areas**

All layers were reprojected to the ETRS89-LAEA grid (EPSG:3035) at 1-km resolution, the reference grid used for environmental reporting in the European Union, so that cell areas are equal by construction. Country boundaries for Spain and Portugal were rasterised onto the same grid. For the representation analysis we used the full CORINE Land Cover 2018 classification rather than the seven natural and semi-natural classes employed as model predictors, since coverage objectives apply to the entire territory. The 100-m product was aggregated to 1 km by majority rule and Level-3 classes were grouped where their individual extent was small, yielding a set of mutually exclusive classes covering 99.8% of the land surface. For each country we computed the proportion of each class and of each 200-m elevation band covered by PAs and by PAs and RLAs combined, and compared both against the 30% coverage objective. The two habitat sets are deliberately different. The seven-class set represents the naturalness gradient and is appropriate as a model predictor, whereas the full classification is required for representation accounting.

To identify which RLAs would contribute most, we built a composite prioritisation index over the RLAs-PAs layer. Cells classified as artificial surfaces, arable land, permanent crops or water bodies were excluded beforehand, since these are not appropriate targets for area-based conservation designation. Four criteria were then used: (i) proximity, the Euclidean distance from each cell to the nearest PAs cell; (ii) bridging capacity, the distance to the nearest PAs cell belonging to a different patch, which identifies cells able to connect otherwise separate units; (iii) representation deficit, derived from the country-specific coverage of the CORINE class and elevation band of the cell, rescaled so that the least represented class and band within each country score highest; and (iv) patch size, the area of the candidate patch. Each criterion was converted to a percentile rank within each country and the four ranks were summed with equal weights. Cell scores were averaged within candidate patches, and patches smaller than 10 $km^2$ were excluded from the ranking. Robustness was assessed under three alternative weightings, each assigning 0.4 to one criterion and 0.2 to the rest, compared using Spearman correlations and top-30 overlap. None of the environmental predictors used in the models of Section 2.6 was reused as a prioritisation criterion, so that the two analyses remain independent.

All spatial processing and the prioritisation analysis were performed in R v4.3.3 (R Core Team, 2021).

## 3. Results

**Quantification of the extent contribution of protected areas and roadless areas by habitat type and elevation gradient**

PAs and RLAs covered comparable portions of the IP (27.0% and 25.1%; Table 1), yet only 8.9% of the territory was shared by both designations, so that RLAs lying outside PAs accounted for almost two thirds of the roadless extent. This low overlap indicates that the two capture largely distinct portions of the landscape.

In terms of land cover, the proportion of territory covered by PAs ranged from 8.2% in Complex cultivation patterns (CC) and 12.6% in Arable land and permanent crops (AR) to 58.8% in Sparsely vegetated and bare areas (SV) and 85.1% in Inland wetlands (WE) (Fig. 2). Among the natural and semi-natural classes, coverage was relatively even, between 28.7% in Mixed forest (MX) and 44.1% in Coniferous forest (CN). Agro-forestry areas (AF) stood out from this pattern: despite being the class with the highest proportion of roadless land of all (44.5%), they were the least covered by PAs of all vegetated semi-natural classes (27.9%). Regarding RLAs alone, the occupied area ranged from 6.5% in CC to 44.5% in AF, followed by Inland wetlands (40.3%) and Sclerophyllous vegetation (SC, 36.9%).

The combined coverage of RLAs and PAs (RLAs∪PAs) exceeded 45% in all natural and semi-natural classes, reaching its maximum in Inland wetlands (87.2%), Sparsely vegetated and bare areas (69.7%) and Sclerophyllous vegetation (62.5%). The contribution of RLAs lying outside PAs (RLAs-PAs) differed markedly among classes (Fig. 2): it was largest in Agro-forestry areas, accounting for 31.1% of the class extent, more than any other class, and lowest in Inland wetlands (2.1%) and Complex cultivation patterns (5.8%). Expressed instead as a proportion of the area already protected, the relative contribution of RLAs to PAs was again highest in Agro-forestry areas (48%) and lowest in Complex cultivation patterns (8.8%) (Fig. S3a). The spatial overlap between the two designations (RLAs∩PAs) remained below 21% of class extent in every class except Inland wetlands, indicating that RLAs and PAs capture distinct, complementary portions of the landscape.

Along the elevation gradient, bands between 1200 and 3000 m were already well represented by PAs, exceeding 50% of occupied extent (Fig. S3b, Fig. S4). The relative contribution of RLAs to PAs varied non-monotonically, from 25.1% in the lowest band to values above 40% between 1400 and 2600 m, dropping to 20.0% in the uppermost band (Fig. S3b). Bands above 2000 m cover less than 1000 km² each, so estimates there should be read with caution. In absolute terms the pattern was reversed: RLAs added up to 19.0 percentage points below 1200 m but less than 7 above 1600 m (Fig. S4), since PAs already cover more than 80% of the terrain there.

Because the 30% coverage objective of the EU Biodiversity Strategy for 2030 and Target 3 of the Kunming-Montreal Global Biodiversity Framework are reported at national level, we assessed the contribution of RLAs separately for Spain and Portugal (Tables 2 and 3). Both countries currently fall below the threshold: PAs cover 27.9% of Spain and 22.3% of Portugal. Incorporating RLAs would raise coverage to 44.9% and 34.3% respectively, so that both would exceed the 30% objective. The aggregate Iberian figure (27.0% to 43.3%) conceals a substantial asymmetry between the two countries, with Portugal 5.6 percentage points further from the target than Spain.

The land-cover picture differs sharply between countries (Table 2). In Spain, all seven strictly natural and semi-natural classes already exceed 30% under the current network. The clearest deficit lies elsewhere: agro-forestry areas, the dehesa and montado systems of the south-west, are covered by PAs over only 30% of their extent in Spain and 21.7% in Portugal, yet they show by far the largest gain from the incorporation of RLAs in both countries (+ 31.4 and + 30.2 percentage points, reaching 61.4% and 51.8% respectively). No other class approaches this contribution. In Portugal, six classes remain below the objective

even when all RLAs are incorporated, including coniferous forest (28.6%) and mixed forest (20.8%); leaving aside artificial surfaces, which are not a target for area-based designation, these delimit the residual gap that roadlessness alone cannot close and that would require active restoration or new designations. Intensively used land behaves as expected: arable land and permanent crops rise from 12.0% to 30.9% in Spain but remain at 29.1% in Portugal, and artificial surfaces stay near 6% in both countries.

Along the elevation gradient, the current network shows a pronounced bias (Fig. S4, Table S3). Across the Iberian Peninsula, PAs cover 15.7% of land below 200 m but 87.0% of land above 2000 m. RLAs deliver their largest contributions precisely in the intermediate bands where the network is weakest, adding between 17.5 and 19.4 percentage points in Spain between 200 and 1200 m, and comparatively little at the highest elevations, which are already saturated. The two countries again diverge at the lower end; in Spain the lowest band remains below the objective after incorporating RLAs (27.1%), whereas in Portugal it just crosses it (31%), although the 400–600 m band there stays below (29.6%). RLAs therefore mitigate, but do not eliminate, the lowland deficit of the conservation network.

Table 2
Coverage of each CORINE Level-3 class (grouped where appropriate) by PAs and by PAs and RLAs combined, reported separately for Spain and Portugal. Gains are expressed in percentage points (pp). The final column indicates whether the 30% objective is met once RLAs are incorporated. Classes are mutually exclusive and jointly cover 99.8% of the terrestrial surface.

| Country | CORINE class | km² | PAs (%) | +RLAs (%) | Gain (pp) | ≥ 30% |
|---|---|---|---|---|---|---|
| Spain | Broadleaved forest | 50,569 | 41.6 | 53.2 | + 11.6 | Yes |
| | Coniferous forest | 48,452 | 46.0 | 59.8 | + 13.8 | Yes |
| | Mixed forest | 11,340 | 36.0 | 55.9 | + 19.9 | Yes |
| | Natural grassland | 31,542 | 38.8 | 58.1 | + 19.3 | Yes |
| | Moors and heathland | 20,308 | 44.4 | 52.9 | + 8.5 | Yes |
| | Sclerophyllous vegetation | 47,193 | 42.2 | 62.6 | + 20.4 | Yes |
| | Transitional woodland-shrub | 20,513 | 36.7 | 58.9 | + 22.2 | Yes |
| | Agro-forestry areas | 25,047 | 29.9 | 61.3 | + 31.4 | Yes |
| | Agriculture with natural vegetation | 10,648 | 18.8 | 33.7 | + 14.8 | Yes |
| | **Complex cultivation patterns** | 17,060 | 7.0 | 13.9 | + 6.9 | **No** |
| | **Pastures** | 10,708 | 18.5 | 28.7 | + 10.2 | **No** |
| | Sparsely vegetated and bare areas | 6,240 | 64.6 | 75.8 | + 11.3 | Yes |
| | Inland wetlands | 901 | 84.0 | 86.6 | + 2.6 | Yes |
| | Arable land and permanent crops | 180,200 | 12.0 | 30.9 | + 19.0 | Yes |
| | Water bodies | 2,706 | 51.9 | 70.0 | + 18.1 | Yes |
| | **Artificial surfaces** | 8,740 | 4.9 | 6.5 | + 1.5 | **No** |
| | **All land** | 493,190 | 27.9 | **44.9** | + 17.0 | Yes |
| Portugal | Broadleaved forest | 8,046 | 24.6 | 41.3 | + 16.7 | Yes |
| | **Coniferous forest** | 3,936 | 19.8 | 28.6 | + 8.8 | **No** |
| | **Mixed forest** | 4,875 | 11.9 | 20.8 | + 8.9 | **No** |
| | Natural grassland | 331 | 32.0 | 44.1 | + 12.1 | Yes |
| | Moors and heathland | 5,035 | 41.6 | 50.4 | + 8.8 | Yes |
| | Sclerophyllous vegetation | 2,442 | 52.5 | 60.8 | + 8.3 | Yes |
| | Transitional woodland-shrub | 16,954 | 22.9 | 35.5 | + 12.6 | Yes |
| | Agro-forestry areas | 8,305 | 21.7 | 51.8 | + 30.2 | Yes |
| | **Agriculture with natural vegetation** | 6,434 | 22.1 | 28.8 | + 6.6 | **No** |
| | **Complex cultivation patterns** | 6,302 | 11.5 | 14.2 | + 2.7 | **No** |
| | Pastures | 1,979 | 22.2 | 42.7 | + 20.5 | Yes |
| | Sparsely vegetated and bare areas | 1,226 | 29.5 | 38.7 | + 9.1 | Yes |
| | Inland wetlands | 213 | 89.7 | 89.7 | + 0.0 | Yes |
| | **Arable land and permanent crops** | 19,026 | 19.2 | 29.1 | + 9.9 | **No** |
| | Water bodies | 688 | 25.1 | 53.3 | + 28.2 | Yes |
| | **Artificial surfaces** | 2,630 | 6.2 | 6.5 | + 0.4 | **No** |
| | **All land** | 88,709 | 22.3 | **34.3** | + 12.0 | Yes |

## Effects of the addition of roadless areas on landscape connectivity

The incorporation of RLAs into the network of PAs (RLAs∪PAs) modified regional landscape connectivity across the three main structural components of connectivity (area, isolation, and aggregation) (Table 3).

In terms of area, adding RLAs to the network of PAs resulted in a substantial expansion of total conservation coverage: Total Area (CA) increased by 60.2%, while the Percentage of Landscape (PLAND) rose by 16.3 percentage points, from 27.02% to 43.29% (Table 3). This expansion was accompanied by a marked reorganisation of patch configuration. The Number of patches (NP) increased from 2218 to 6231 (+ 180.9%), while Mean patch area (AREA_MN)

decreased by 43.0%. Crucially, the Standard deviation of patch area (AREA_SD) increased by 25.6%, indicating that patch sizes became more heterogeneous rather than more uniform. This combination is informative: the incorporation of RLAs simultaneously added many small patches at the periphery of the network and merged several previously separate PAs into larger continuous units, so that both the lower and the upper tail of the patch-size distribution expanded. RLAs therefore do not perform a single structural role; some act as stepping stones between existing patches while others consolidate the core of the network.

Table 3
Landscape metrics comparing the spatial pattern of PAs in isolation with PAs and roadless areas combined (RLAs∪PAs) for the Iberian Peninsula, computed at class level on a 1-km equal-area grid (ETRS89-LAEA). Symbols + and - indicate an increase or decrease, respectively, caused by the addition of RLAs. For PLAND the change is expressed in percentage points.

| Landscape metrics | Units | PAs | RLAs∪PAs | +/- | Change (%) |
|---|---|---|---|---|---|
| *Area* | | | | | |
| CA | ha | 15721100.0 | 25193000.0 | + | 60.2 |
| PLAND | % | 27.02 | 43.29 | + | 16.3 |
| NP | - | 2204 | 6231 | + | + 181.8 |
| AREA_MN | ha | 7137.96 | 4058.17 | - | -43.1 |
| AREA_SD | | 57486.57 | 72247.35 | + | + 25.7 |
| *Isolation* | | | | | |
| ENN_MN | m | 3388.43 | 2596.63 | - | -23.4 |
| ENN_SD | | 2242.90 | 1048.20 | - | -53.3 |
| *Aggregation* | | | | | |
| CONTAG | % | 41.19 | 21.05 | - | -48.9 |

CA: Total (Class) Area; PLAND: Percentage of landscape; NP: Number of patches; AREA_MN: Mean patch area; AREA_SD: Standard deviation in patch area; ENN_MN: Mean of the Euclidean Nearest Neighbour Distance Distribution; ENN_SD: Standard deviation of the Euclidean Nearest Neighbour Distance Distribution; CONTAG: Contagion index.

In contrast, the inclusion of RLAs reduced patch isolation. The Mean Euclidean Nearest Neighbour Distance (ENN_MN) decreased by 23.4% (from 3385.43 m to 2592.63 m), while the Standard deviation (ENN_SD) declined by nearly 54% (from 2272.9 m to 1047.2 m) (Table 3). These changes indicate shorter and more homogeneous distances among patches, suggesting that the expanded conservation network may facilitate movement among habitat patches and improve structural connectivity across the landscape.

Finally, the Contagion index decreased from 41.21 to 21.05 (– 48.9%; Table 3), indicating that the conserved class became more broadly distributed across the landscape.

Overall, the inclusion of RLAs reshaped the conservation network into a denser but more spatially dispersed configuration, characterised by a higher number of smaller patches that were closer to each other across the landscape.

## Environmental drivers of protected and roadless areas

Comparing predictor sets within each response variable showed that PAs and RLAs are governed by different environmental drivers (Table S4). For PAs, the topographic model (TOPO) was the best supported and achieved the highest coefficient of discrimination of any model fitted in this study (D = 0.11), with the habitat and biodiversity-landscape models progressively less supported. For RLAs, by contrast, the biodiversity and landscape connectivity model (BIOLAND) was the best supported, and the same ordering was recovered when RLAs lying outside PAs were modelled in isolation. The composite BEST model did not outperform the single-theme models for any response variable. Notably, the coefficient of discrimination remained low for RLAs (D ≈ 0.01–0.02) irrespective of the predictor set used, whereas it reached 0.11 for PAs. Rather than indicating poor model quality, this contrast is itself informative: PAs occupy a comparatively narrow and predictable environmental profile, whereas RLAs are distributed across the full breadth of the environmental space and are therefore not captured by any single set of environmental descriptors. Full model comparison statistics are given in Table S4.

## Environmental differences between protected and roadless areas

The contribution achieved by each predictor in the $PAs_{BEST}$ and $RLAs_{BEST}$ models is summarised in Fig. S5. Terrain Ruggedness (TRI) held the highest contribution in both models, but nearly twice as high for $PAs_{BEST}$ as for $RLAs_{BEST}$ (65% vs 35%). In $PAs_{BEST}$, TRI was followed by the Shannon Diversity Index (SHDI, 17%), Broadleaved forest (BL, 13%) and the Topographic Position Index (TPI, 5%), with the Euclidean Nearest-Neighbour distance (ENNm) and Moors and heathland (MO) contributing marginally (1.3% and 0.1%, respectively). In $RLAs_{BEST}$, by contrast, the two topographic attributes dominated jointly (TRI 35% and TPI 29.5%), while SHDI (15.7%) and MO (13.8%) showed more balanced scores and BL and ENNm contributed less than 5% each (4.9% and 1.2%).

Both PAs and RLAs were related primarily to topographic and landscape attributes, but through contrasting associations. The response curves of the three most contributing predictors in each model (Fig. S6) show that the probability of occurrence increases with terrain roughness for PAs, whereas for RLAs it

declines as terrain becomes steeper and rougher. Consistently, the distribution of TRI values differed markedly between the two, with PAs occupying rougher ground than RLAs (Fig. 3). RLAs were also associated with a broader range of relief positions, spanning valley bottoms, flat areas and ridges, although the two designations did not differ appreciably in their TPI distributions (Fig. 3).

Landscape and biodiversity attributes behaved differently. SHDI followed a similar pattern in both models (Fig. S6), with values increasing as the probability of occurrence decreased, indicating that areas less likely to be protected were associated with higher landscape heterogeneity; accordingly, the two designations did not differ in their SHDI distributions (Fig. 3). The same applied to MO and to Species Richness (SRCH), despite MO being among the largest contributors to $RLAs_{BEST}$. By contrast, BL, Endemicity (ENDM) and ENNm all differed clearly between the two, with PAs showing higher BL and ENDM values and RLAs greater ENNm, that is, more isolated patches (Fig. 3). Given the sample size, the absence of a detectable difference for MO, SHDI, SRCH and TPI is itself informative: with approximately 583,000 grid cells even small differences would be detected, so these predictors can be considered to have essentially equivalent distributions in PAs and RLAs.

## Spatial prioritisation of roadless areas

Not all RLAs contribute equally to closing the gaps identified above. Of the 94719 $km^2$ of RLAs lying outside PAs, we first excluded cells classified as artificial surfaces, intensively cultivated land or water bodies, which are not appropriate targets for area-based designation, leaving 57803 $km^2$ of eligible land (61.0% of the total). These were ranked using a composite index combining four criteria computed separately for each country (Section 2.7): proximity to existing PAs, capacity to bridge distinct PAs patches, relative representation deficit of the CORINE class and elevation band occupied, and patch size. After discarding patches smaller than 10 $km^2$, 1180 patches in Spain and 231 in Portugal were retained for ranking.

The ranking proved robust to the weighting scheme. Spearman correlations between the equal-weight index and three alternative weightings ranged from 0.938 to 0.961, and between 22 and 25 of the 30 highest-ranked patches were common to all schemes in both countries. The lowest agreement occurred for Spain under the representation-weighted scheme, which is expected given that most Spanish classes are comparatively well represented and the representation criterion therefore discriminates weakly there.

Priority patches were markedly concentrated in the south-western quadrant of the Peninsula, in the agro-silvo-pastoral landscapes of Extremadura, western Andalusia and the Portuguese Alentejo (Fig. 4, Table 4). Agro-forestry areas were the dominant class in 35 of the 60 highest-ranked patches across both countries, far ahead of any other class, so that the prioritisation converges independently on the same systems identified as most deficient in Table 2. Incorporating the 300 highest-ranked patches would add 13229 $km^2$ in Spain, equivalent to 2.7% of its territory, raising national coverage from 27.9% to 30.6% and thereby meeting the 30% objective. Portugal, by contrast, cannot reach the threshold through roadlessness alone; its 231 eligible patches larger than 10 km2 amount to 5796 km2 in total and would raise coverage only to 28.9%.

Table 4
Fifteen highest-ranked roadless patches in each country under the equal-weight index. The dominant CORINE class is the most frequent class within the patch. Centroids are given in geographic coordinates (WGS84).

| Country | ID | km² | Index | Dominant CORINE class | Elev. (m) | Lat, Lon |
|---|---|---|---|---|---|---|
| Spain | 4199 | 76 | 0.809 | Agro-forestry areas | 373 | 39.29, -6.41 |
| | 4842 | 123 | 0.807 | Agro-forestry areas | 185 | 38.62, -7.25 |
| | 3550 | 237 | 0.794 | Sclerophyllous vegetation | 305 | 39.76, -6.66 |
| | 5107 | 92 | 0.792 | Agro-forestry areas | 276 | 38.45, -7.09 |
| | 3348 | 54 | 0.786 | Agro-forestry areas | 308 | 39.98, -6.70 |
| | 3859 | 120 | 0.783 | Transitional woodland-shrub | 364 | 39.61, -6.06 |
| | 5364 | 63 | 0.779 | Agro-forestry areas | 528 | 38.44, -5.89 |
| | 5629 | 57 | 0.775 | Transitional woodland-shrub | 357 | 38.01, -7.11 |
| | 4115 | 47 | 0.774 | Agro-forestry areas | 642 | 39.49, -5.72 |
| | 2824 | 48 | 0.768 | Agriculture with natural vegetation | 252 | 41.24, 0.11 |
| | 2177 | 45 | 0.765 | Agro-forestry areas | 756 | 40.88, -6.35 |
| | 4592 | 81 | 0.765 | Agro-forestry areas | 252 | 39.02, -6.44 |
| | 3643 | 53 | 0.761 | Agro-forestry areas | 320 | 39.76, -6.34 |
| | 4483 | 26 | 0.760 | Agro-forestry areas | 366 | 39.26, -5.63 |
| | 5703 | 164 | 0.760 | Transitional woodland-shrub | 313 | 37.91, -7.13 |
| Portugal | 4909 | 27 | 0.820 | Agro-forestry areas | 180 | 38.46, -8.12 |
| | 4679 | 32 | 0.813 | Mixed forest | 81 | 38.56, -8.57 |
| | 4812 | 24 | 0.794 | Mixed forest | 42 | 38.44, -8.59 |
| | 5714 | 250 | 0.788 | Agro-forestry areas | 196 | 37.85, -7.31 |
| | 3397 | 87 | 0.771 | Agro-forestry areas | 201 | 39.79, -7.35 |
| | 5847 | 32 | 0.765 | Agro-forestry areas | 237 | 37.63, -8.12 |
| | 5030 | 35 | 0.762 | Coniferous forest | 95 | 38.26, -8.61 |
| | 4386 | 15 | 0.755 | Broadleaved forest | 19 | 38.76, -8.86 |
| | 4994 | 11 | 0.751 | Coniferous forest | 54 | 38.29, -8.68 |
| | 5049 | 32 | 0.738 | Transitional woodland-shrub | 90 | 38.22, -8.70 |
| | 5887 | 24 | 0.735 | Agro-forestry areas | 165 | 37.74, -7.50 |
| | 4731 | 43 | 0.729 | Broadleaved forest | 57 | 38.49, -8.66 |
| | 5010 | 63 | 0.717 | Agro-forestry areas | 173 | 38.33, -8.13 |
| | 3326 | 63 | 0.713 | Transitional woodland-shrub | 309 | 39.94, -6.89 |
| | 4639 | 15 | 0.690 | Mixed forest | 82 | 38.62, -8.51 |

## 4. Discussion

Previous studies have highlighted the potential of RLAs to complement networks of PAs by quantifying the spatial gains obtained after their incorporation (Crist et al. 2005; Psaralexi et al. 2017; Chen et al. 2024). However, the value of RLAs for conservation planning depends not only on their extent, but also on how they are distributed across the landscape, how their environmental characteristics compare with those of existing PAs, and how much of the gap towards binding coverage targets they can actually close. Here we combined descriptive spatial assessment, modelling and spatial prioritisation to address these three questions for the Iberian Peninsula. Our results show that RLAs are spatially complementary to PAs, substantially expand the potential conservation network, and are governed by different environmental drivers.

## Roadless areas as a complementary resource for expanding protected areas

RLAs occupy an extent comparable to that of the current network of PAs in the Iberian Peninsula, yet less than 9% of the territory is shared by both designations. This low overlap indicates that RLAs and PAs capture largely distinct portions of the landscape, and their combination increases the total area

under conservation by more than 60%. The expansion was not spatially uniform. The largest single gain fell on agro-forestry systems, which rose by more than 30 percentage points in both countries, followed by transitional woodland-shrub and sclerophyllous vegetation. Agro-forestry landscapes are simultaneously the most roadless and among the least protected land in the Peninsula, a combination that makes them the clearest representation gap the current network leaves open.

A similar pattern emerged along the elevation gradient, where RLAs contributed mainly to low and mid-elevation ranges (Strittholt and Dellasala 2001; Crist et al. 2005). This is consistent with prior evidence that PAs are biased towards remote, rugged or less productive land, leaving lowland and mid-elevation landscapes under-represented (Joppa and Pfaff 2009). Unlike heavily pressured coasts or high-elevation areas already largely protected, mid-elevation lands such as Sierra Morena in southern Spain or the Alentejo in southern Portugal remain productive yet sparsely populated and largely unaltered by resource extraction or road construction. Taken together, our results indicate that RLAs address representativeness gaps in both dimensions, land cover and elevation, and constitute a strategic resource for expanding conservation coverage.

## Landscape connectivity implications of incorporating roadless areas into protected areas

The incorporation of RLAs modified landscape structure across the three connectivity components. Total conserved area and the percentage of landscape under conservation both increased substantially. At the same time, the number of patches rose sharply while mean patch area fell, and, crucially, the variability of patch size increased rather than decreased. This combination is informative, since RLAs simultaneously added many small patches at the periphery of the network and merged previously separate PAs into larger continuous units, expanding both tails of the patch-size distribution. RLAs therefore do not perform a single structural role; some act as stepping stones while others consolidate the core of the network.

The isolation metrics support this argument. Despite the rise in patch number, Euclidean nearest-neighbour distances decreased and became less variable. Patches became more numerous yet closer to one another, indicating improved structural connectivity. Although our analyses do not assess functional connectivity for particular taxa, the reduced inter-patch distances are consistent with RLAs facilitating dispersal for species that use smaller patches as stepping stones (Baum et al. 2004; Herrera et al. 2017; Diniz et al. 2021; Wang et al. 2021). The decrease in Contagion does not contradict this, since it is expected on compositional grounds when the conserved class rises from 27.0% to 43.3% of the landscape.

## Environmental similarities and differences between roadless areas and protected areas

The modelling results support the idea that RLAs and PAs are related but not equivalent conservation entities, and they specify in what sense. Within each response variable the best-supported predictor set differed between the two. PAs were best explained by topography, consistent with the well-documented tendency for protected areas to be designated on rugged, inaccessible and economically marginal land (Joppa and Pfaff 2009), whereas RLAs were best explained by landscape structure and biodiversity. The two designations are therefore not merely differently located; they are governed by different processes. The consistently low coefficient of discrimination obtained for RLAs under every predictor set further indicates that roadless conditions persist across a much wider portion of the environmental space than that occupied by the current network of PAs, providing quantitative support for a complementarity argument that has so far rested mainly on descriptive grounds (Crist et al. 2005; Psaralexi et al. 2017; Talty et al. 2020).

This asymmetry is corroborated independently by the representation analysis. Terrain ruggedness was the dominant predictor in both models but contributed nearly twice as much for PAs, and the elevation profile shows the same signal from a different direction (PAs cover 15.7% of land below 200 m but 87.0% above 2000 m). RLAs, by contrast, spanned valleys, flat areas and ridges alike, so that roadless conditions can be maintained across a wider range of topographic contexts than those currently represented within PAs (Strittholt and Dellasala 2001; Crist et al. 2005).

The biodiversity predictors point to both similarity and differentiation: species richness did not differ appreciably, whereas endemicity was higher in PAs, consistent with the targeted designation of many protected areas in locations harbouring range-restricted biota. RLAs should therefore be understood as a strategic complement to, not a replacement for, existing protected areas.

## Implications for conservation planning and future practices

Despite their global expansion, PAs alone have proven insufficient to halt biodiversity loss or ensure ecosystem connectivity, often reflecting limited management capacity and spatial biases rather than true ecological representativeness (Pimm et al. 2018; Visconti et al. 2019; Elleason et al. 2021) (Elleason et al., 2021; Pimm et al., 2018; Visconti et al., 2019). Because PAs tend to be sited on land of low economic interest while biodiversity threats operate both inside and outside them (Margules and Pressey 2000; Nenzén and Araújo 2011; Mazaris et al. 2013), expanding the network remains a contested issue. RLAs have long been proposed as an efficient route for such expansion, given their conservation value as relatively undisturbed land (Selva et al. 2011; Ibisch et al. 2016; Chen et al. 2024; Hoffmann et al. 2024), and some countries already recognise them formally (Nie and Barns 2012), yet in Europe the initiatives remain few (IENE 2015) and although EU-wide assessments have recently begun to fill this gap (Testolin et al. 2026).

Our results suggest two non-exclusive pathways, and allow both to be quantified. First, RLAs adjacent to existing PAs could be prioritised to enlarge current sites; in the EU this is administratively the least demanding route, since it can proceed as a perimeter extension of existing Natura 2000 sites rather than as a new designation. Second, isolated RLAs of high environmental value could be recognised as Other Effective Area-based Conservation Measures (OECMs), a category designed for areas delivering in-situ conservation outcomes without formal protected-area status and which counts towards the 30% objective. The prioritisation presented here indicates that Spain could meet that objective by incorporating some 13229 km² of roadless land, equivalent to 2.7% of its

territory, a far smaller commitment than the headline figure of all RLAs would suggest. Portugal, by contrast, cannot reach the threshold through roadlessness alone, since its full set of eligible patches would raise coverage only to 29.0%, and would need to combine designation with active restoration, particularly in coniferous and mixed forest.

A third implication follows from the concentration of priority patches in agro-forestry systems. Dehesas and montados are High Nature Value farmland whose evergreen *Quercus* formations are listed as habitat type 6310 in Annex I of the Habitats Directive, yet they are among the least protected classes in both countries. Because they depend on continued extensive grazing, designation alone would be insufficient and would need to be paired with instruments that sustain management, such as agri-environmental payments under the Common Agricultural Policy, rather than with strict protection. Implementation is also institutionally uneven. In Spain the competence rests largely with the autonomous communities, whereas in Portugal it is centralised in the Instituto da Conservação da Natureza e das Florestas. Recognising RLAs further creates friction with infrastructure policy, in particular with the extension of the trans-European transport network and with road investment financed under national recovery and resilience plans, since both operate on precisely the low-elevation land where RLAs contribute most. Finally, formal protection may attract new forms of human use, including tourism, potentially eroding the very qualities that make these areas valuable; any strategy should therefore be accompanied by clear management objectives and measures to maintain low levels of disturbance and road expansion.

Finally, several limitations should be considered when interpreting our findings. First, our analyses were conducted at a spatial resolution of 1 km, which is suitable for regional assessments but may overlook fine-scale ecological processes. Second, the connectivity metrics used here describe structural rather than functional connectivity, and therefore species-specific responses to the observed landscape patterns remain uncertain. Third, the models were fitted to approximately 583,000 contiguous grid cells, and residual spatial autocorrelation was not explicitly modelled; with a sample of this size, statistical significance is largely a function of sample size. We therefore base our interpretation on the direction and magnitude of the differences shown in Fig. 3, and on the shape of the response curves, rather than on p-values alone. Fourth, the reduction in the Contagion index after adding RLAs is expected on purely compositional grounds, since the proportion of the landscape assigned to the conserved class rises from 27.0% to 43.3%; we therefore do not interpret it as a loss of ecological aggregation. Fifth, the definition of RLAs is based on roadlessness and low traffic intensity, but other dimensions of human pressure may also influence their ecological condition. Sixth, and related, estimates of roadless extent are strongly definition-dependent: using OpenStreetMap with stricter filtering criteria, Testolin et al. (2026) report substantially lower roadless extents for the same countries, so the figures presented here should be read as conditional on the roadless dataset used. Future work should assess RLAs at multiple spatial scales, incorporate functional connectivity analyses for different taxa, and evaluate how roadless areas perform under alternative conservation or management scenarios.

## Conclusion

Roadless areas (RLAs) are a substantial yet under-recognised component of conservation landscapes. Our results show that RLAs are spatially complementary to existing protected areas (PAs), substantially increasing the area under conservation and contributing to a more extensive and spatially distributed network of conserved habitats. At the same time, the incorporation of RLAs modifies landscape structure by increasing the number of smaller patches and reducing distances among them, suggesting a denser configuration of conservation areas across the landscape. In addition, RLAs share several environmental characteristics with PAs, while also incorporating distinct habitat types and elevation ranges that are underrepresented in current protected area systems. This complementarity highlights their potential to improve both the spatial extent and ecological representativeness of conservation networks. These findings suggest that RLAs could play an important role as a strategic complement to PAs, particularly in regions where conservation systems are spatially biased or incomplete. However, their contribution should be evaluated in relation to landscape context, ecological characteristics, and their role within broader connectivity patterns. Improved mapping, monitoring, and ecological assessment of RLAs will be essential to support their integration into planning frameworks. Explicitly considering RLAs within conservation strategies may strengthen biodiversity conservation by expanding and reorganising existing networks of PAs while maintaining low levels of human disturbance.

## Declarations

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**Author contribution**

SA-C: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Software, Supervision, Validation, Visualization, Writing – original draft, Writing – review and editing. OP-B: Data curation, Formal analysis, Writing – original draft. RV: Visualization, Writing – review and editing. NS: Conceptualization, Formal analysis, Investigation, Methodology, Visualization, Writing – review and editing. All authors critically contributed to reviewing, editing and revising the manuscript and gave final approval for publication.

**Data availability**

All primary datasets used in this study are publicly available: the global roadless areas map and its conservation status assessment from http://www.roadless.online/; the Nationally Designated Areas (CDDA) and Natura 2000 inventories from the European Environment Agency (https://www.eea.europa.eu/); CORINE Land Cover 2018 from the Copernicus Land Monitoring Service; the European digital elevation model (EU-DEM); and

the terrestrial biodiversity indicators from the World Bank global database. The derived raster layers, the landscape metrics computed for each response variable and the R code used for all analyses are deposited in Zenodo (DOI: 10.5281/zenodo.22129198).

**Competing interests**

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Figures

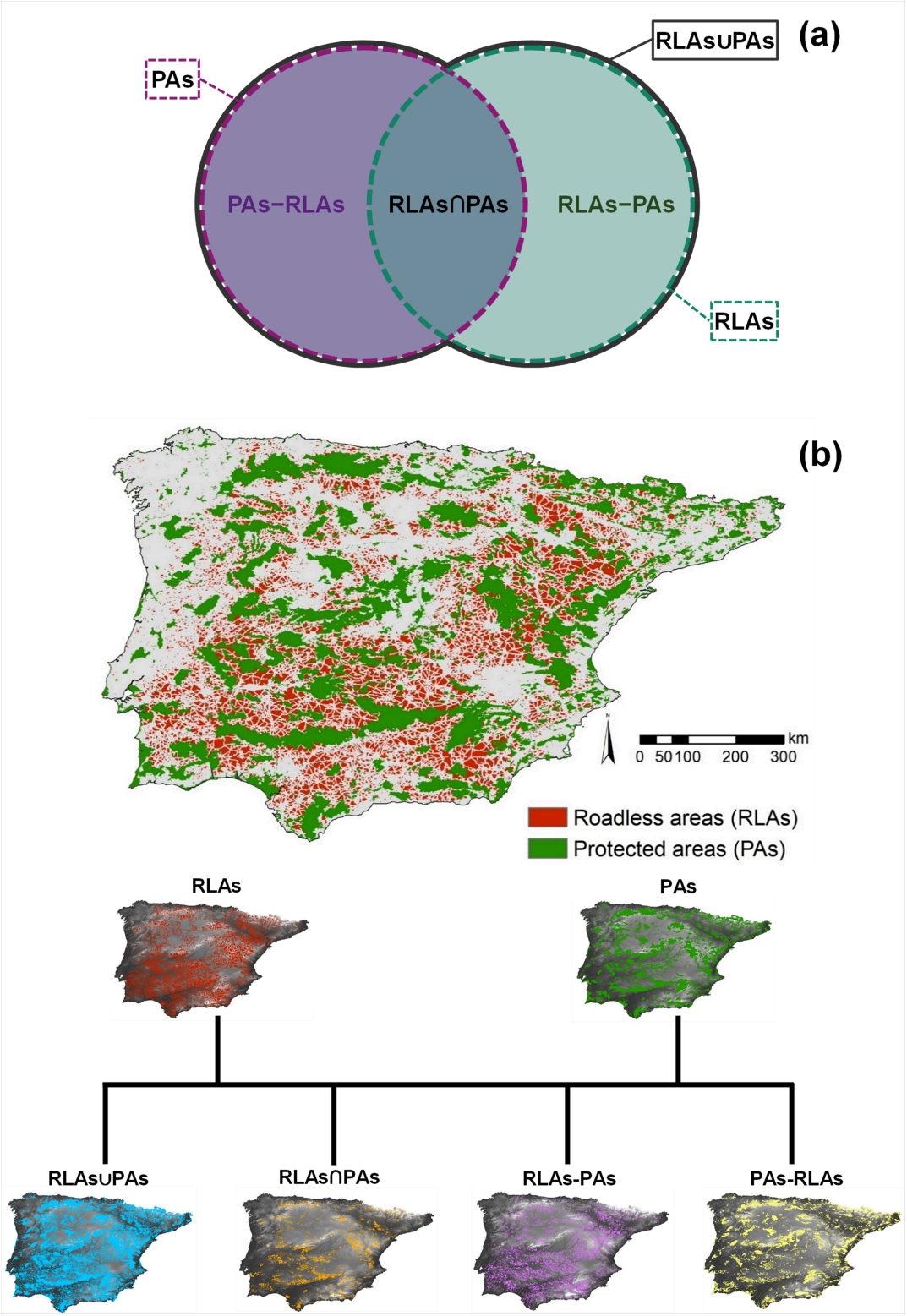



**Figure 1**

(a) Roadless and protected areas scheme; (b) Map of the Iberian Peninsula with the distribution of roadless areas (RLAs) and protected areas (PAs) at 1 × 1 km pixel size, and the union of RLAs and PAs (RLAs∪PAs), the intersection (RLAs∩PAs), the RLAs in isolation (RLAs-PAs), and the PAs in isolation (PAs-RLAs).

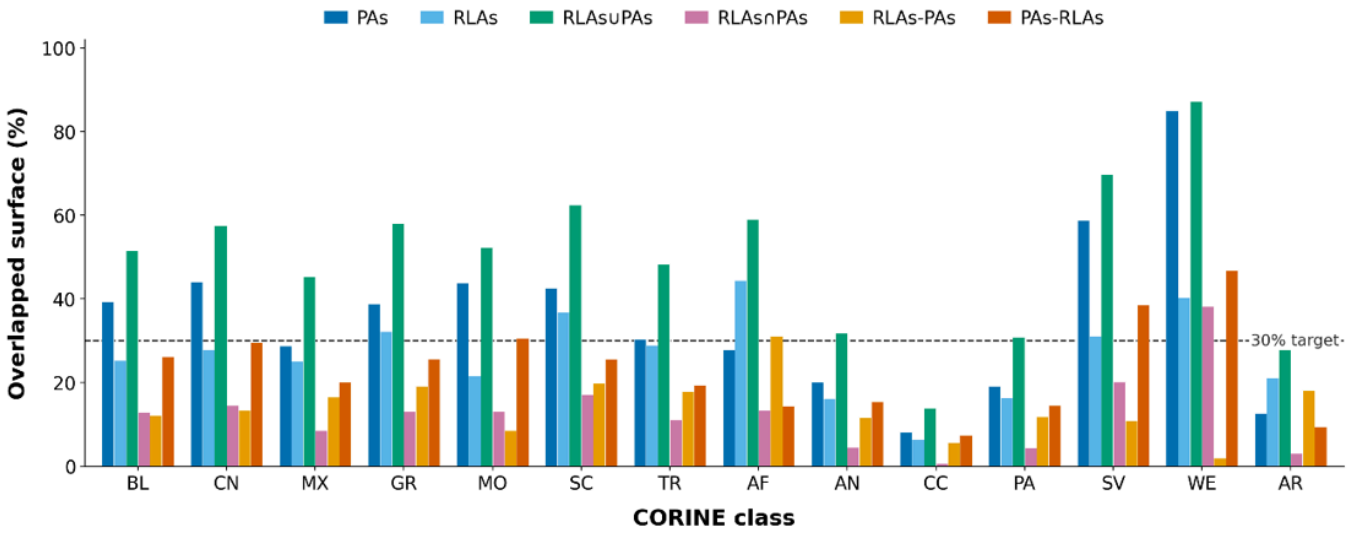



**Figure 2**

Overlapped surface (%) between each CORINE class in the Iberian Peninsula and PAs, roadless areas (RLAs), RLAs and PAs merged (RLAs∪PAs), intersected (RLAs∩PAs), and RLAs (RLAs-PAs) and PAs (PAs-RLAs) in isolation. Classes: Broad-leaved forest (BL), Coniferous forest (CN), Mixed forest (MX), Natural grasslands (GR), Moors and heathlands (MO), Sclerophyllous vegetation (SC), Transitional shrubs (TR), Agro-forestry areas (AF), Agriculture with natural vegetation (AN), Complex cultivation patterns (CC), Pastures (PA), Sparsely vegetated and bare areas (SV), Inland wetlands (WE), Arable land and permanent crops (AR). The dashed line marks the 30% coverage objective.

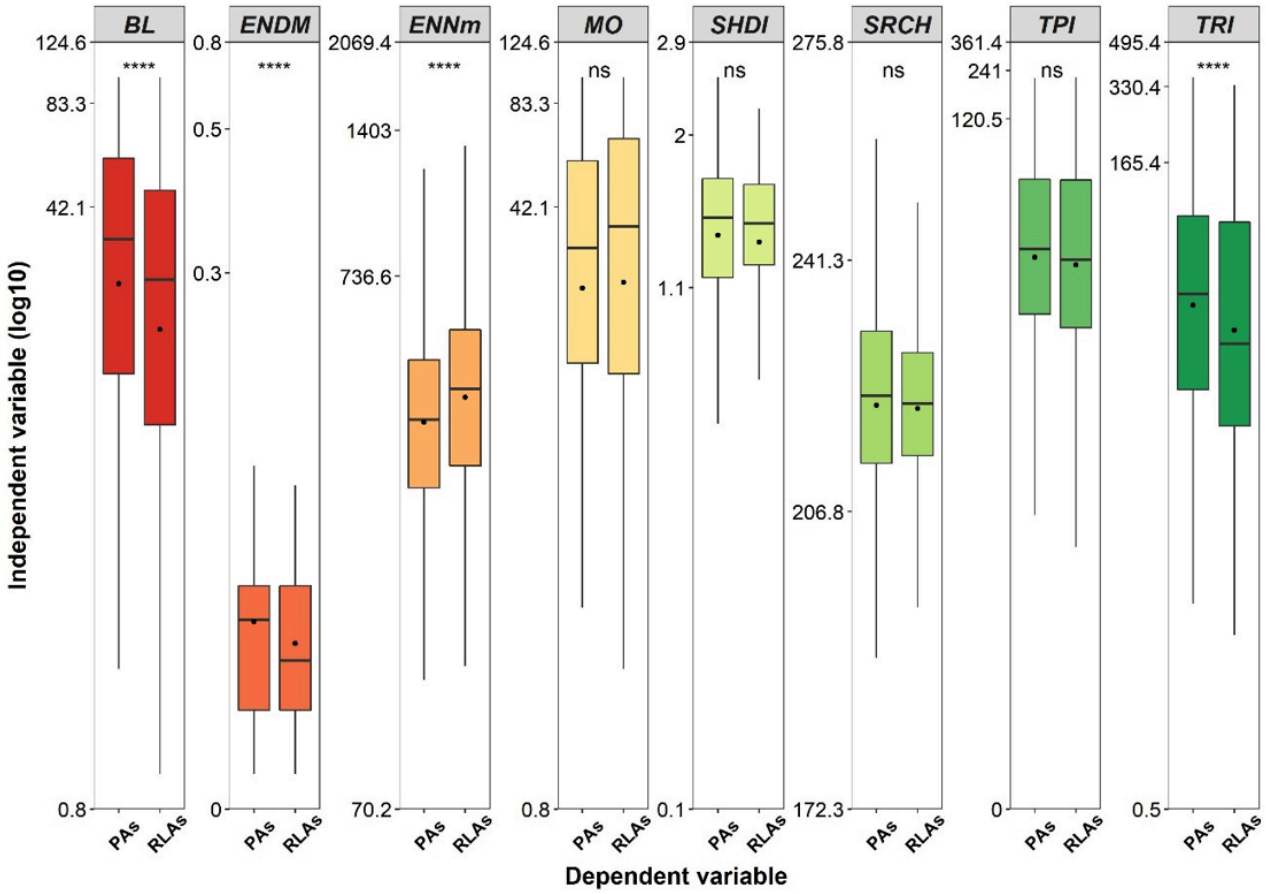


**Figure 3**

Distribution of the most contributing predictors in PAs and roadless areas (RLAs), compared with t-tests. Boxes show the interquartile range, the horizontal line the median and the point the mean; whiskers extend to 1.5 times the interquartile range. Values are displayed on a $\log_{10}$ scale. Predictors: Broadleaved forest (BL), Endemicity (ENDM), Euclidean Nearest-Neighbour distance (ENNm), Moors and heathland (MO), Shannon Diversity Index (SHDI), Species Richness (SRCH), Topographic Position Index (TPI) and Terrain Ruggedness Index (TRI). ENDM and SRCH were retained here because they were among the most contributing predictors in the biodiversity and landscape connectivity (BIOLAND) model sets, although they were not selected for the BEST models. Significance: **** $p < 0.0001$; ns, not significant. Given the sample size (approximately 583000 grid cells), significance should be read alongside the magnitude of the differences shown.

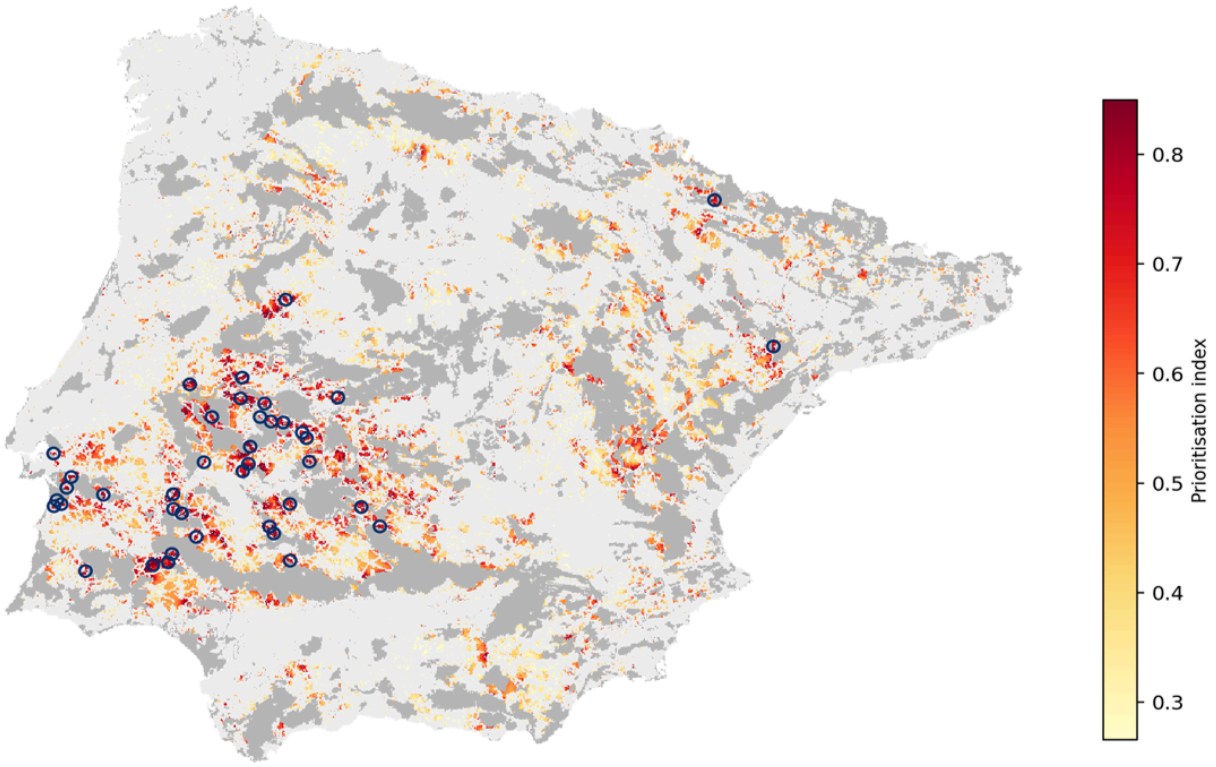



**Figure 4**

Priority roadless areas for expanding the network of PAs in Spain and Portugal. Light grey: land. Mid grey: existing PAs. Colour scale: prioritisation index for eligible RLAs lying outside PAs. Circles indicate the 40 highest-ranked patches.

## Supplementary Files

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- Supplementarymaterial.rar