A robust and comprehensive indicator for the economic valuation of threatened biodiversity

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Abstract

We present a new framework for the selection of indicators for the standardized valuation of threatened biodiversity. Criteria for the selection of indicators were based on the concepts of threat status and regional responsibility for species preservation. Indicators were all species included in the Annexes of the European Birds and Habitats Directives inhabiting the valued systems in the target area, plus the endemic species either 'Critically Endangered' or 'Endangered' not (yet) covered by the Directives. Criteria were applied to the forest of Andalusia (southern Spain), a region of 87,268 km², with 44,460 km² (ca. 54%) classified as forest habitats. The final list included 224 species: 81 plants, 76 birds, 31 mammals, 22 arthropods, six reptiles, five amphibians, and three mollusks. Indicators were combined by means of an index of relative conservation value accounting for differences in threat status, sensitivity, functional role, and amount of knowledge. Distribution maps of each species at the 1 km x 1 km scale required for economic valuations were available for 108 species (48%). The remaining maps were downscaled from published 10 km x 10 km UTM maps, using knowledge on habitat requirements (preferred vegetation types) and altitudinal ranges. Fine-scale maps were overlapped with the 159,764 forest patches recognized in the Andalusia Forest Map and weighed by relative conservation values of each species. As expected, conservation values were larger in protected than in unprotected areas, and larger in dehesas (a kind of seminatural habitat of high conservation value) than in other forests. The logic of the procedure can be applied to any region of the world as it provides objective criteria based on two relevant concepts for analyzing the existence value of threatened biodiversity, i.e. threat level and responsibility for preservation, and methods to account for differences among species in sensitivity, functional role and relevant knowledge. Overall, the proposed framework overcomes key shortcomings for indicators’ selection criteria, allowing its standardized use for economic valuations of endangered biodiversity.

Key words: Biodiversity valuation, threat status, sensitivity to disturbance, functional role, degree of knowledge.
Introduction

Awareness on the causes and consequences of biodiversity loss is rapidly increasing (Sala et al. 2000, Butchart et al. 2010), and the importance of preserving this scarce public good will increase in future political agendas (TEEB 2010, EC 2011). Development of tools and indicators to help assess the way in which programs and policies will affect this biodiversity are increasingly important and timely. The economic value of environmental products derived from threatened biodiversity is a key aspect that/whose status and evolution should be monitored; however, it is one of the most difficult to quantify both in physical and monetary units (Pearce & Moran 1994).

The valuation of threatened biodiversity involves the estimation of societal preferences in the context of simulated markets, whose design requires relevant and accurate indicators of this biodiversity. These indicators must be based on the physical measurement of the distribution over space and time of unique biological entities (usually species) whose persistence is threatened. The physical measurement of these indicators must be subject to an economic assessment, in this case as passive consumption within a coherent evaluation system able to integrate the existence value of threatened species. Indicators of the risk of biodiversity loss must therefore be based on measurements of the distribution of threatened species in a given geographical area at a given time, and they must be capable of capturing its variation over space and time.

The threatened biodiversity indicators used so far have been based on locally well-known groups or organisms, mostly selected groups of invertebrates, plants and some groups of vertebrates such as birds (Sidding et al. 2016), by explicitly or implicitly assuming that they will be representative of all threatened biodiversity (Nelson et al. 2009, Tallis et al. 2009, Caro 2010).
However, this assumption is questionable, because different groups of organisms tend to respond differently to the pressures that cause their decline or extinction (Lawton & May 1995, Thuiller et al. 2011). In fact, use of well-known groups as indicators was more related to data availability than to its putative indicator character (Simberloff 1998, Caro 2010, Lindenmayer and Likens 2011). Further, each species within a given indicator group are usually considered as equivalent, so that single integrative indicators are number of species, or numbers weighted by their relative abundance if known (i.e. diversity indices), in the cases where more than one indicator species are considered (Sidding et al. 2016). Nevertheless, species may differ in its indicator value of the conservation status of a given locality or system due to, at least, three main traits: 1) threat status (presence of more endangered species would indicate a higher conservation value for the territory holding them; Díaz 2009, Caro 2010); 2) sensitivity to main threats (sensitive species would be better indicators; Díaz 2009, Sidding et al. 2016); 3) functional role (presence of keystone species would indicate a better conservation status; Simberloff 1998, Díaz 2009). Further, availability of relevant information can also be important for either selecting or weighting indicators, as the presence or absence of less-known species would be less informative of the conservation value than presence or absence of the best-know species.

We develop a new framework of selection of indicators for the economic valuation of threatened biodiversity that addresses these limitations. The framework was based on the concepts of the threat level of all species found in a territory or system, as well as on the regional responsibilities for species preservation, sensitivity to disturbances and functional roles of species in ecosystems, as well as knowledge on species distribution, roles, sensitivity
and trends. We develop and applied this new framework for indicator selection in the forests of Andalusia (southern Spain), but as a general framework it can be adapted and applied in other systems and regions worldwide. The framework relies on the broadest-scale legislation available, the European Birds and Habitats Directives in this case, complemented with the most threatened species endemic to the region of interest (Andalusia) not (yet) covered by the broad-scale legislation. This general procedure has two main advantages: 1) it grounds the selection criteria on the regional responsibility for species conservation rather than on the availability of information and 2) it partly ensured this availability for the status, distribution, trends, threats and functional roles of the species listed, as both the European Directives and regional conservation laws oblige to gather this kind of data for the species listed and to update it regularly (EU 1979, 1992, BOE 2003).

Integration of threatened biodiversity valuations also involves its estimation at the spatial scales relevant for other economic valuations. This scale corresponds to the median size of Andalusian forest states in our case (tens-to-hundreds hectares; Campos et al. 2017). We derived distribution maps of the species listed as indicators at this scale using available maps, either raw or downscaled if necessary. Combination of maps with relative weights of indicator species according to threat, sensitivity, functional role and knowledge will produce maps of relative conservation value that can be integrated at several scales, from individual farms to the whole Andalusian forests. We illustrate the performance of this method by comparing the conservation values of protected and unprotected areas, as well as the values of dehesas, a kind of savanna-like European semi-natural habitat of high conservation value (Díaz et al. 1997, 2013, Díaz and Pulido 2009), with the
values of the natural forests dehesas are derived from (oak forests; Díaz et al. 1997) and the values of other forest, shrubland or grassland habitats. We expected a) that relative conservation values should be larger in protected areas, and larger in the National than in the Natural Parks, and b) that dehesas should reach higher conservation values than the oak forests they derived from (Díaz et al. 2013).

Methods

Study area

Selection and valuation frameworks for biodiversity indicators were developed within the large-scale project RECAMAN (Campos et al. 2017; www.recaman.es). The project goal was to produce a spatially-explicit extended ecosystem accounting system including four private activities (forestry, hunting, residential and private amenity) and six public activities (mushroom collection, carbon storage, water regulation, public recreation, landscape and threatened biodiversity). The system was applied to the forest of Andalusia, an autonomous region located in southern Spain. The region extends over 87,268 km², with 44,460 km² (ca. 54%) classified as forest habitats including forests, woodlands, scrublands and grasslands (Junta de Andalucía 2011).

List of indicators

The selection of indicators started with the lists of threatened species included in the Annexes of the European Birds and Habitats Directives (EU 1979,
In a first step, we eliminated species whose distribution areas do not include Andalusia, as well as species linked to non-forest habitats (urban, agricultural, freshwater and marine) on the basis of the information provided by the most recent national red books (Martí and del Moral 2003, Bañares et al. 2004, Pleguezuelos et al. 2004, Verdú and Galante 2006, 2008, Palomo et al. 2008, Palomino et al. 2012). We then completed this reduced regional list with the species endemic to Andalusia that are either 'Critically Endangered' or 'Endangered' according to regional red books (Blanca et al. 1999, Franco and Rodríguez 2001, Barea-Azcón et al. 2008), and are not (yet) covered by the Directives.

Relative conservation value of indicators

We developed a specific methodology for weighting the list of indicator species which, starting from the list of threatened species, incorporates functional aspects in the analysis of the conservation value of Andalusian forest (Noss 1990). Relative indexes were assigned to each species according to their degree of threat, potential response to four factors of disturbance considered key in the context of forest management (fragmentation, fire, grazing and silvicultural practices), functional role in forest systems, and the quality of available information on their distribution, abundance and trends.

Weighting factors

For threat status we used the categories developed by the IUCN (EW: extinct in the wild, now reintroduced; CR: Critically Endangered; EN: Endangered; VU: Vulnerable; LR: Low Risk, combining the IUCN categories NT –Near
Threatened-, LC –Least Concern- and DD-Data Deficient), which synthesize quantitative and/or qualitative information on population size, range size, and recent trends in population and range (Mace et al. 2008). Updated IUCN categories were extracted from the most recent regional red books (Blanca et al. 1999, Franco and Rodríguez 2001, Barea-Azcón et al. 2008). Species protected by European Directives but not explicitly listed in Andalusia due to its low regional threat level were considered as Near Threatened (NT) if rare at the European scale or of Least Concern (LC) if abundant or increasing.

Disturbances considered relevant for the analysis of the conservation value of the Andalusian forests were fragmentation, wildfires, grazing (by domestic and wild herbivores) and silvicultural practices such as cuts, felling, removal of dead wood or old trees, pesticide applications, etc. (Herrera 2004). Categories for each disturbance were (1) demonstrated effects; (2) effects deduced from the biological characteristics of species; and (3) demonstrated absence of effects or no information.

Functional roles of species in forest systems were obtained or deduced from the biological information reviewed in red books. Ecosystem engineers are species whose presence or activity affects the flows of matter and energy in the system. In temperate forests, the main ecosystem engineers are the dominant trees (Manning et al. 2006, Diaz et al. 2013). Keystone species directly or indirectly affect the species integrated into their food web through predation, dispersal, competition and/or predation. Typical keystones are top predators, medium and large herbivores, animal dispersers of the dominant trees, and scrub facilitators of tree recruitment (Simberloff 1998; review in Diaz et al. 2015). The remaining species were considered as subordinate to
engineers and keystones. Categories were thus (1) engineers, (2) keystones and (3) subordinate species.

Availability and quality of information on population size, distribution and trends was assessed from the literature review and the information available from the Environment Department of the regional government of Andalusia. Four categories were established according to the type of information available: (1) Regular and detailed information gathered in the frame of monitoring programs in which the total population size and range are determined and global estimates of demographic parameters (birth, mortality and movements) are obtained for predicting future evolution; (2) regular information based on periodic and standardized censuses of the total size of species population, throughout the whole known range of distribution, or in a sample that can be extrapolated to the total area; (3) regular reports on species conservation status with a uniform but non-standardized methodology throughout their distribution range; and (4) review of partial works, such as Atlas and Red Books of widely distributed species with methodological difficulties for their census. Details on categories assigned to each species can be found in Díaz et al. (2015) and in Appendix 1.

Combination of weighting factors

Criteria for assigning relative weights to categories within each factor considered are essentially arbitrary in the absence of solid and accurate information on the relative relevance of the different factors (Díaz et al. 2001). The same is true for combination methods (additive, multiplicative). For this reason, we chose the simplest criterion: linear scales to assign relative weights
of each factor considered, and additive combination of relative indices to calculate the overall index. Seven factors were then combined, ranging from three to five categories (see above), with a maximum value of 18. Effects of arbitrariness on the scaling and combination criteria were tested by correlating simpler indices with two more complex indices. The first used additive weights obtained after rescaling each factor to values ranging from 0 to 1, dividing the initial values by the maximum value of the index for each factor, thus avoiding that factors with more levels weighed more in the final value of the index. The second index resulted from the combination of factors obtained using principal components analysis; principal components are independent from each other, refer to the same scale and synthesize the co-variation patterns of the original weighting factors (Pärtel et al. 2005).

**Spatially-explicit combination of indicators**

We derived distribution maps of the species listed as indicators at the scale required for economic valuation from the available cartography. Sources for these maps were (1) regional distribution maps of threatened species provided by the regional Administration; (2) the Spanish Vertebrates Database (MMA 2000); (3) red books of Andalusia invertebrates (Barea-Azcón et al. 2008) and plants (Blanca et al. 1999); (4) the databases of the Anthos project (Anthos 2011) for vascular plants not included in the red books; (5) the national butterfly atlas for diurnal Lepidoptera (García-Barros et al. 2004); and (6) the atlas of wintering birds (Palomino et al. 2012). Recent updates extracted from a comprehensive literature review using species’ names as keywords in Internet searches (Google Scholar for the years after the publication of red
books, accessed along 2014) were also used for some species (details in Díaz et al. 2015).

We used directly official maps at the 1 km x 1 km UTM grid or at finer scales maintained by the regional government. The remaining maps were published at the 10 km x 10 km UTM grid scale. In these cases, we used the information available on the habitat requirements of these species (preferred vegetation types) and their altitudinal ranges to estimate what areas within these 10 km x 10 km squares were most likely occupied by each species. These methods, although less precise than those based on direct censuses, would, however, produce more realistic estimates than wide-scale presence-absence maps (Araújo et al. 2005). Maps were downscaled by overlapping distribution maps with the types of habitats occupied by each species through Geographic Information Systems (GIS). We updated species distribution maps from the most recent Atlas and from similar projects. We obtained the types of forest land occupied by each species in Andalusia from a comprehensive literature review on species requirements (Appendix 1; Díaz et al. 2015), after grouping the forest land types defined in the digital maps available (Andalusia vegetation map) into a smaller number of categories. Thus, we grouped the 72 forest land types (plus five additional categories of land use or cover) that were recognized in the vegetation map into 16 types according to dominant species (e.g., pines *Pinus* spp., deciduous or evergreen *Quercus* spp. oaks etc.) and woodland structure (closed forest or open woodland; Suppl. Table 1; Díaz et al. 2015). Downscaling by overlapping distribution and vegetation maps was done separately for each species and Andalusia province; we then merged the eight provincial maps into a single regional map per species. We excluded agricultural, urban and freshwater areas.
We used distribution maps at the 1 km x 1 km UTM grid scale to ascertain the presence-absence of each of the 224 indicator species in each of the 159,764 forest patches of differing size recognized in the Andalusia Forest Map by GIS overlapping. These presence-absence patterns, together with estimates of the size of the distribution area of each species obtained by summing the size of all suitable and likely occupied patches, were the physical bases for the estimation of the economic value of threatened biodiversity in the forest of Andalusia.

Comparative examples: Protected vs. unprotected areas, and dehesas vs. other forest types

Maps of the two Andalusian National Parks (Sierra Nevada and Doñana) and the 22 Natural Parks were obtained from the Internet. Using SIG, we assessed whether each of 159,764 patches defined above were inside a National or a Natural Park, or outside, Dehesas are European semi-natural habitats of high conservation value (Díaz et al. 1997, 2013, Díaz and Pulido 2009, Campos et al. 2013). Dehesas are open oak *Quercus* spp. woodlands where trees grow scattered over a grassland, scrub, or cultivated understory following a savanna-like configuration. Rather than being planted, trees were retained when the original oak forests was opened to create dehesas by means of tree pruning and felling, shrub removal and ground ploughing in long rotations (Díaz et al. 1997, Moreno and Pulido 2009). Andalusian forest patches were classified as dehesas if they were occupied by open oak woodland with 5-75% tree cover and as oak forests if tree cover was larger (Campos and Caparrós 2016). *Quercus* forests have more than 75% tree cover. The remaining patches were classified as other forests types, or as shrubland or grassland.
Comparison among forest types and protection categories were made by means of GLMs on raw values, and on log-transformed values of raw values divided by patch size to account for effects of this variable. Differences among categories were tested by means of *a posteriori* Tukey tests.

**Results**

The final list included 224 species: 81 plants, 76 birds, 31 mammals, 22 arthropods, six reptiles, five amphibians, and three mollusks (Appendix 1). Most species were included in the Endangered (EN; 67 species) and Vulnerable (VU; 55 species) categories. The list of indicators also included 10 species Critically Endangered (CR) and two Extinct in the Wild and reintroduced (RE). More than 60 species were sensitive to forestry practices or grazing (67% and 62%, respectively), whereas less than 50% were sensitive to wildfires (42%) or fragmentation (30%). Only 18% species had keystone functional roles, and only 20 species (9%) had a good level of knowledge on its abundance, range and trends, whereas the information available for most species was partial or incomplete (61%; Appendix 1).

The values of the index after rescaling the linear weights assigned to each factor were highly correlated with the initial magnitudes, either if groups of factors were not grouped or if the factors related to the effect of disturbances were grouped together (*r* = 0.94, $R^2 = 88\%$, $p << 0.0001$ and *r* = 0.95, $R^2 = 90\%$, $p << 0.0001$, *n* = 224). Similarly, the initial magnitudes correlated strongly with the first principal component extracted from the analysis of the co-variation pattern of the values of the different factors considered (*r* = 0.94,
R² = 89%, p << 0.0001). Additive combination and linear scaling gave thus similar results than more complex combinations.

The values of the conservation index varied between 2 and 14, with modal values of 5-6 (Fig. 1). Mean values varied according to regional threat status (Fig. 2). Critically endangered and reintroduced species has the highest values, endangered and data deficient species intermediate values, and vulnerable and low risk species the lowest mean values.

Fine-scale maps maintained by the regional conservation authorities were available for 108 species (48%). Out of the remaining 124, 37 occupy agricultural habitats as well as open forests, shrublands and grasslands; 27 prefer shrublands and grasslands; 27 prefer open woodlands; nine occupy riverine forests; and 15 are narrow-range specialist of specific forest types (Table 1). Conservation values of vegetation patches based on these indicator species varied between 1 and 346 (mean: 148; Fig. 3). Total values were significantly correlated with values based on the 108 species with fine-scale maps available (rS=0.166, N=159764, P=0.000), but variance explained was very low (r²=2.75%). Hence, distribution of the best-known species largely undervalued the distribution of endangered biodiversity.

Peak conservation values were found in the better-preserved areas of Andalusia, namely the National Parks of Sierra Nevada and Doñana, and the Natural Parks of Alcornocales and Gibraltar area, mountain ranges along the Mediterranean coast, Gata Cape, Cazorla and Segura Mountains, and parks along the Sierra Morena mountain range in the northern border of Andalusia (Fig. 3). Conservation values were larger in areas within National than within Natural Parks, and larger than outside protected areas (Fig. 4), both absolutely (F₂,₁59764=851.99, P<<0.0000) and relative to patch sizes (F₂,₁59764=849.69,
Conservation values were larger in dehesas than in the *Quercus* forests dehesas were created from, and larger than in other forests, whose conservation value was however larger than those of *Quercus* forests (Fig. 5). Values for shrubland and grasslands were much larger than values for dehesas and forests (Fig. 5). Trends were similar for both absolute values ($F_{3, 159764} = 113650.05, P << 0.0000$) and values relative to patch sizes ($F_{3, 159764} = 2742.06, P << 0.0000$).

**Discussion**

The method developed here generated a list of manageable size including all species of conservation concern for a country-size, species rich European territory. Spatially-explicit combination of the relative conservation value associated to each species by means of fine-scale maps, weighed by differences among species in threat status, sensitivity to disturbance, functional role and availability of information, allowed to obtain a ‘conservation map’ that captured differences in conservation value both among protected and unprotected areas within the target territory and among habitat types derived from specific land uses. Relative conservation values were correlated with the economic valuation of the existence value of endangered biodiversity at the spatial scale required for integration of commercial and non-commercial valuations, allowing us to incorporate threatened biodiversity valuations into standardized accounting systems for the total economic value of ecosystems (Campos et al. 2018). Hence, the method developed here met the requirements established by several authors and international agencies for use in rigorous assessments of threatened
biodiversity, and can be used for the economic valuation of biodiversity through non-market valuation techniques (Nijkamp et al. 2008, Carson 2012).

Selection of indicator species was based on threat status and regional responsibility to ensure its long-term persistence, rather than on regional knowledge or popularity (Sidding et al. 2016). Nevertheless, endangered species may differ in its indicator value due to, at least, differences in threat status, sensitivity to disturbance or functional role in ecosystems (Simberloff 1998, Caro 2010, Sidding et al. 2016). Explicit consideration of these differences by means of simple categorization and combination criteria allowed us to combine data for all species. Simple categorization and combination criteria (linear scaling and additive combination; Díaz et al. 2001) performed equally than methods more complicated or sophisticated, including principal component analyses of patterns of co-variation among traits (Pärtel et al. 2005). Hence, the method developed here resulted robust for combining information for all relevant species.

Spatially-explicit combination of these indicator weights requires accurate presence-absence maps of all species. These maps, and their regular update, are mandatory for the species selected under both regional and continental conservation laws (EU 1979, 1992, BOE 2003). However, the fine-scale accurate maps needed for economic valuation (Campos et al. 2017) seem unfeasible to obtain for many species, especially the rarest and more difficult to detect. In fact, less than half of the selected indicator species had suitable maps maintained by the regional government.

Large-scale maps based on the 10 km x 10 km UTM grid have been produced for all threatened species compiling the information available during production of national and regional red books. Downscaling these maps using
species’ habitat requirements and altitudinal ranges allowed us to estimate what areas within these 10 km x 10 km squares were most likely occupied by each species. This method is less precise than direct censuses, but would produce more realistic estimates than wide-scale presence-absence maps (Araújo et al. 2005). Use of actual presence-absence data avoid the overestimation of distribution areas usually obtained with niche envelope modeling approaches (Watling et al. 2013), whereas use of habitat requirements within occupied areas would include suitable patches for each species independently of actual occupation (dark biodiversity approach; Pärtel et al. 2011, Lewis et al. 2016). Overall, selection of species whose updated distribution is mandatory, simple downscaling methods to apply to these maps if necessary, and weights based on the quality of information available allowed us to develop fine-scale maps of the conservation value of endangered biodiversity useful for its economic valuation. Final values were correlated with values based on the species with most closely followed by the regional government, but high scatter implied very low variance explained by this correlation. Conservation values based on the subset of the best-known species were thus inaccurate to estimate values for all relevant species.

The procedure developed here can be applied to any region of the European Union if the process starts with the lists of species of the Birds and Habitat Directives, or to any other region of the world, starting with the most appropriate legislation on threatened species and habitats (for example, the US Endangered Species Act). Methods for downscaling, weighting and combining species’ maps can also be applied to any region, although their accuracy may vary according to the degree of detail of regional knowledge on the geographic distribution, threat status and habitat requirements of the selected
species. In any case, the procedure provides objective methods based on two relevant concepts for analyzing the existence value of threatened biodiversity, i.e. threat level and responsibility for preservation, thus obviating subjective choices of indicators for threatened biodiversity based on the degree of local or regional knowledge. Objective criteria for species’ selection avoided arbitrary increases of conservation values based on adding more and more species of conservation concern (Bateman et al. 2011). Further, avoidance of effects of degree of knowledge or popularity on conservation action (Díaz et al. 2018) implies that valuation of existence values based on these lists would approach conceptually the estimation of the intrinsic values of endangered biodiversity (Batavia and Nelson 2017). Overall, the method proposed overcomes several key shortcomings of usual selection criteria of indicators (Sidding et al. 2016), allowing its standardized use for economic valuations of endangered biodiversity.

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Supplementary material

**Appendix 1** Species selected as indicators of the conservation value of Andalusian forests. Scientific name; taxonomic group (AM: amphibians; AR: arthropods; B: birds; ML: mollusks; MM: mammals; P: plants; R: reptiles); Annex in Birds or Habitat Directives; threat status in the most recent Andalusian red books; relative values associated to status, sensitivity to forest fragmentation, wildfires, grazing and forestry practices, functional role and degree of knowledge, and the sum of these values into a conservation value; habitat types selected and elevation range; and main data sources, are given for each of the 224 indicator species.

**Supplementary Table 1.** Equivalence between the vegetation categories considered in the Andalusian vegetation map (both in Spanish and in English) and the vegetation categories considered in this study according to the main vegetation requirements of the indicator species. See Appendix 1 for requirements and sources of data, and Díaz et al. (2015) for details.
Figure legends

Fig. 1. Frequency distribution of the relative conservation value of the 224 species selected as indicators of the conservation status of Andalusian forests.

Fig. 2. Mean (±SE) relative conservation values of species according to regional threat status extracted from Andalusian Red Data Books. Lines on the top of the figure indicate homogeneous groups according to Tukey tests after a one-way ANOVA (F$_{6, 217}$=10.21, P<<0.0001). EW: extinct in the wild, now reintroduced; CR: Critically Endangered; EN: Endangered; DD: Data Deficient; VU: Vulnerable; NT: Near Threatened; LC: Least Concern.

Fig. 3. Map of the values of the relative conservation index of Andalusian forests. Boundaries of protected areas are also indicated: the National Parks of Doñana and Sierra Nevada, the Natural Parks surrounding them, and the remaining 22 Andalusian Natural Parks (1: Cádiz Bay; 2: Barbate Marshes; 3: Strait of Gibraltar; 4: Alcornocales; 5: Grazalema; 6: Sierra de las Nieves; 7: Montes de Málaga; 8: Tejeda and Almijara; 9: Huétor; 10: Sierra de Baza; 11: Gata Cape; 12: Sierra María-Los Vélez; 13: Castril; 14: Cazorla, Segura and Las Villas; 15: Mágina; 16: Despeñaperros; 17: Andújar; 18: Cardeña and Montoro; 19: Sierras Subbéticas; 20: Hornachuelos; 21: Sierra Norte de Sevilla; 22: Aracena). Location of Andalusia in western Europe is shown in the inlet.
**Fig. 4.** Mean (±SE) relative conservation values of forests patches located inside protected areas (National or Natural Parks) or outside them (Unprotected), both in absolute values (left) and relative to patch sizes, log-transformed (right). Y-axis scaling is the same as in Fig. 5 for comparison. Differences among all means were statistically significant (α=0.05) according to a Tukey test. N= 5606, 49487 and 104669 for National Parks, Natural Parks, and Unprotected areas, respectively.

**Fig. 5.** Mean (±SE) relative conservation values of dehesa, *Quercus* forests, other forest types, and shrubland and grassland patches, both in absolute values (left) and relative to patch sizes, log-transformed (right). Y-axis scaling is the same as in Fig. 4 for comparison. Differences among all means were statistically significant (α=0.05) according to a Tukey test. N= 24255, 28363, 60623 and 46523 for dehesa, *Quercus* forests, other forest types, and shrubland and grassland, respectively.
Figure 1

Histogram showing the number of species for different conservation values.
Figure 2

![Graphic representation of conservation value ± SE for different regional threat statuses. The x-axis represents regional threat statuses (EW, CR, EN, DD, VU, NT, LC), and the y-axis represents conservation value ± SE. The data points are shown for each status, indicating the variability in conservation value across different statuses.](image-url)
Figure 3
Figure 4

The figure shows a comparison of conservation values across different categories: National Park, Natural Park, and Unprotected. The conservation values are presented both in raw and log-transformed (log10) forms. Error bars indicate the standard error.
Figure 5

The figure shows a comparison of conservation values across different land cover types: DEHESA, Quercus forests, OTHER forests, and SHRUBLAND and GRASSLAND. The conservation values are plotted on a logarithmic scale (log10), and error bars represent the standard error (SE) of the conservation value. The values range from approximately 200 to 300 on the y-axis, with a logarithmic scale on the x-axis ranging from -3.5 to -1.5.