Journal of Environmental Management 395 (2025) 127794

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

# Unsound renewable energy source development threatens an umbrella species in a Mediterranean biodiversity hotspot

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ARTICLE INFO

*Keywords:*
Biodiversity conservation
Greece
Large carnivore conservation
Photovoltaic impact
Spatial planning
Windfarm impact
Brown bear

ABSTRACT

Harnessing the power of Renewable Energy Sources is considered a viable solution for reducing anthropogenic carbon emissions that drive climate change, and it is an integral element of current climate policies. However, the development of utility-scale Renewable Energy Sources is now outpacing that of distributed renewable energy and is directly linked to habitat loss and other negative impacts on biodiversity. Using an umbrella species, the brown bear (*Ursus arctos*), as a case study, we assessed whether the development of Renewable Energy Sources, as implemented currently in Greece and other countries worldwide, is compatible with the conservation of a biodiversity hotspot. Our results indicate that the spatial development of Renewable Energy Sources in Greece has not paid the necessary attention regarding a critical umbrella species and its habitats, as potentially more than 50 % of the suitable habitat will be negatively affected; if unchallenged, the combined effects of habitat loss and increased anthropogenic activity of the current Renewable Energy Sources development will threaten brown bears and biodiversity in general in Greece. To address this, the development of Renewable Energy Sources should adopt a spatial planning approach aligned with biodiversity conservation priorities, strengthen environmental assessment quality controls for project approval, and implement systematic monitoring of ongoing projects to minimise the impacts of Renewable Energy Sources on biodiversity. Our methodology and conclusions are transferable to other species and locations. They are therefore of global significance, given the continuous decline in biodiversity and the rapid development of Renewable Energy Sources worldwide.

## 1. Introduction

At the dawn of the 21st century, climate change and biodiversity loss are two anthropogenic crises that threaten our subsistence and well-being (Díaz et al., 2019). Harnessing the power of Renewable Energy Sources (RES; e.g., wind and solar power) is considered a viable solution for reducing the anthropogenic carbon emissions that are driving climate change and is an integral element of current climate policies (e.g., Paris Agreement). However, utility-scale RES development is currently outpacing distributed renewable energy (IEA, 2024) and is directly linked to land loss and habitat fragmentation (Kati et al., 2021), while having other negative impacts on wildlife species (Cryan et al., 2014), the full extent of which is not yet thoroughly understood. Adverse effects on biodiversity have been recorded, amongst others, from the development of photovoltaic, wind energy (i.e., terrestrial and offshore), bioenergy and hydropower projects (He et al., 2024; Santangeli et al., 2016). Most research on the effects of RES development on biodiversity has focused on the impact on flying vertebrates, while preliminary studies on large carnivores highlight the adverse effects of increased anthropogenic disturbance, the direct impacts of land changes through habitat loss and fragmentation, and the potential effects on population viability (Ferrão da Costa et al., 2018; Sirén et al., 2017). The negative impacts of land changes on habitats and species are particularly relevant to Europe and the Mediterranean region in particular, where such threats are now considered more pressing than those posed by climate change (WWF et al., 2020), even though the Mediterranean region and its ecosystems are particularly prone to the effects of climate change due to increased variability in rainfall and altered key ecosystem drivers (Abbott and Le Maitre, 2010; Newbold et al., 2020). Therefore, protecting and restoring European habitats and species has been at the

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https://doi.org/10.1016/j.jenvman.2025.127794
Received 10 May 2025; Received in revised form 7 October 2025; Accepted 24 October 2025

centre of the European Union's environmental policy (European Commission, 2021, 2022). Considering the strong interactions between climate, land use and biodiversity (Ritchie et al., 2020), we have now arrived at a crossroad, often termed as the green-green dilemma (Voigt et al., 2019), where current policies and actions appear to counteract each other, requiring decisions on how to move forward in the most sensible way to preserve biodiversity.

The brown bear (*Ursus arctos*) is a large carnivore that may well illustrate the conservation implications of the green-green dilemma in Europe and other areas across its wide distribution; the species is non-territorial and an opportunistic omnivore that utilises numerous food sources over vast regions and various temporal scales. Therefore, bears have large spatial requirements regarding habitat availability and connectivity; consequently, they may serve as an excellent umbrella species (Simberloff, 1999). Additionally, due to their role as a keystone species, brown bears may be particularly crucial for habitat conservation (Johnson et al., 2017), as they have a disproportionately large impact on the balance and functioning of ecosystems (e.g., facilitating seed dispersal or nutrient cycling). Their overall behavioural plasticity and adaptability have allowed them to survive in various environmental contexts, and in the case of Europe, in 10 populations throughout the continent. Most of these populations have shown encouraging signs of recovery; however, some populations, particularly in southern Europe (e.g., Italy, Spain, Greece), remain small and endangered (Swenson et al., 2020). Legal protection of bears in Europe is regulated through several legal frameworks; for members of the European Union (EU), however, the "Habitats Directive" is the most relevant and ambitious for species of conservation priority (Linnell and Boitani, 2024). The Directive dictates that EU States need to pursue the Favourable Conservation Status of bears, while ensuring that "*The natural range of the species is neither being reduced nor is likely to be reduced for the foreseeable future*", and that "*There is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis*" (European Community, 1992). These commitments are implemented *in situ* primarily through the pan-European "Natura 2000" (N2K) network of protected areas, which has a wide-ranging "Umbrella effect" when it comes to conserving biodiversity (Linnell and Boitani, 2024).

Within this conservation setting, Greece, a global biodiversity hotspot (Myers et al., 2000), exemplifies the green-green dilemma, as brown bears here inhabit the southern edge of the species' range and are therefore an essential element of European biodiversity. Bears in Greece have recently shown signs of range, genetic, and demographic recovery (Karamanlidis et al., 2021); however, they remain endangered, primarily due to the increase in anthropogenic activities (De Gabriel Hernando et al., 2021), and need systematic conservation actions. At the same time, Greece has set ambitious goals for the development of RES within its national climate policy. Benefiting from the fact that most of the country is public land and its high wind and solar potential, Greece has been favouring the development of windfarms and photovoltaic projects at an unprecedented scale, even in N2Ks. The country has already reached 44 % of its 2030 national goal for wind harnessing (Kati et al., 2021). RES development, however, can have a negative impact on bears at both local (i.e., at the construction site) and distant scales (De Gabriel Hernando et al., 2020; Ferrão da Costa et al., 2018). Direct adverse effects may include habitat loss, habitat fragmentation, land-use change, and an overall increase in human activity. Increased human activity has a profound negative impact on bear behaviour (Hertel et al., 2025). The direct adverse effects of RES development may be exacerbated by the secondary effects of other RES-associated works (e.g., access roads, disposal areas) or the synergistic, cumulative impacts of RES development with other human activities. All these potential adverse effects raise concerns about the compatibility of current RES development efforts with Greece's commitments and goals to protect its biodiversity.

This study aims to assess whether utility-scale RES development, as it is currently being implemented in terms of extension and siting, is compatible with the conservation of a biodiversity hotspot. For this, we used brown bears in Greece as an essential element of European and Mediterranean biodiversity in a case study, in which we: 1) Tested the null hypothesis that the spatial development of RES is currently taking into account habitat suitability for brown bears in Greece (i.e., avoiding areas with higher brown bear habitat suitability); 2) Evaluated the potential impacts of RES development on critical bear habitat. Given the continuous decline in biodiversity and the rapid development of RES worldwide (Rehbein et al., 2020), the methodology developed and the results of our case study are transferable to other species and locations, and are therefore of global significance.

## 2. Methods

### 2.1. Study area and data sources

The study was carried out throughout the range of the brown bear in continental Greece (39°35′ N, 21°50′ E; Fig. 1), excluding the Peloponnese, where bears have not been recorded. We used brown bear presence data collected between 2004 and 2024 within the framework of the "Hellenic Bear Register", a project focusing on the monitoring of brown bears in Greece (Karamanlidis et al., 2008); https://www.hellenicbearregister.com/). The data used (N = 12,051) met the C1 (i.e., undisputable evidence based on hard facts) or C2 criteria (i.e., reliable evidence evaluated by a trained expert) (Molinari-Jobin et al., 2012) that have been adopted by the Large Carnivore Initiative for Europe (LCIE; (Kaczensky et al., 2024). We retained only presence data with a location error <1 km and removed all the duplicates (i.e., records of the species in the same 1 $km^2$ cell). The presence-only dataset obtained (Fig. 1) consisted of the centroids of the 1 $km^2$ cells where the species was recorded (N = 2373 cells) and was used as training data for further habitat suitability modelling.

We compiled a dataset (i.e., polygon layers) on current RES projects in Greece from the geoportal of the Hellenic Regulatory Authority for Energy, Waste and Water (www.geo.rae.gr) (Fig. 1). Within our study area, as of December 2024, this dataset included 1395 wind power (i.e., 140 projects under evaluation, 100 with an installation licence, 1000 with a production licence and 155 with a licence to operate) and 4430 photovoltaic projects (i.e., 714 with an installation licence, 3452 with a production licence and 264 with a licence to operate). The dataset also included 2246 and 4242 rejected wind power and photovoltaic projects, respectively. All spatial data were projected to the European Terrestrial Reference System 1989 (ETRS89) in the European Lambert Azimuthal Equal Area Projection. We generated Kernel density maps of RES projects using the kernel density estimation tool in ArcGIS 10.5 (Esri, 2016), with the default bandwidth computed according to Silverman's rule of thumb. We rescaled to represent project density per square kilometre. Subsequent spatial analyses (e.g., buffer, intersection, overlay, reclassification) and map production were performed in R (R Core Team, 2025), using the software packages "sf" (Pebesma, 2018), "terra" (Hijmans, 2022) and "spatstat" (Baddeley et al., 2016).

### 2.2. RES spatial development in relation to bear habitat suitability

To assess habitat suitability for brown bears in Greece, we employed an inhomogeneous Point Process Model (PPM) approach, using the "ppmlasso" package (Renner and Warton, 2013) and following the guidelines of Bonnet Lebrun et al. (2019) and Warton and Shepherd (2010). In a PPM, the observed presences of the target species are treated as points generated by an underlying spatial process, and the model relates the local intensity of points (i.e., the relative likelihood or expected number of records per cell) to predictor variables (Warton and Shepherd, 2010). This method eliminates the need to generate pseudo-absences (i.e., dummy absences). It enables the inclusion of variables that account for uneven sampling effort and observer bias (e.g., higher reporting in areas with higher observer presence). This

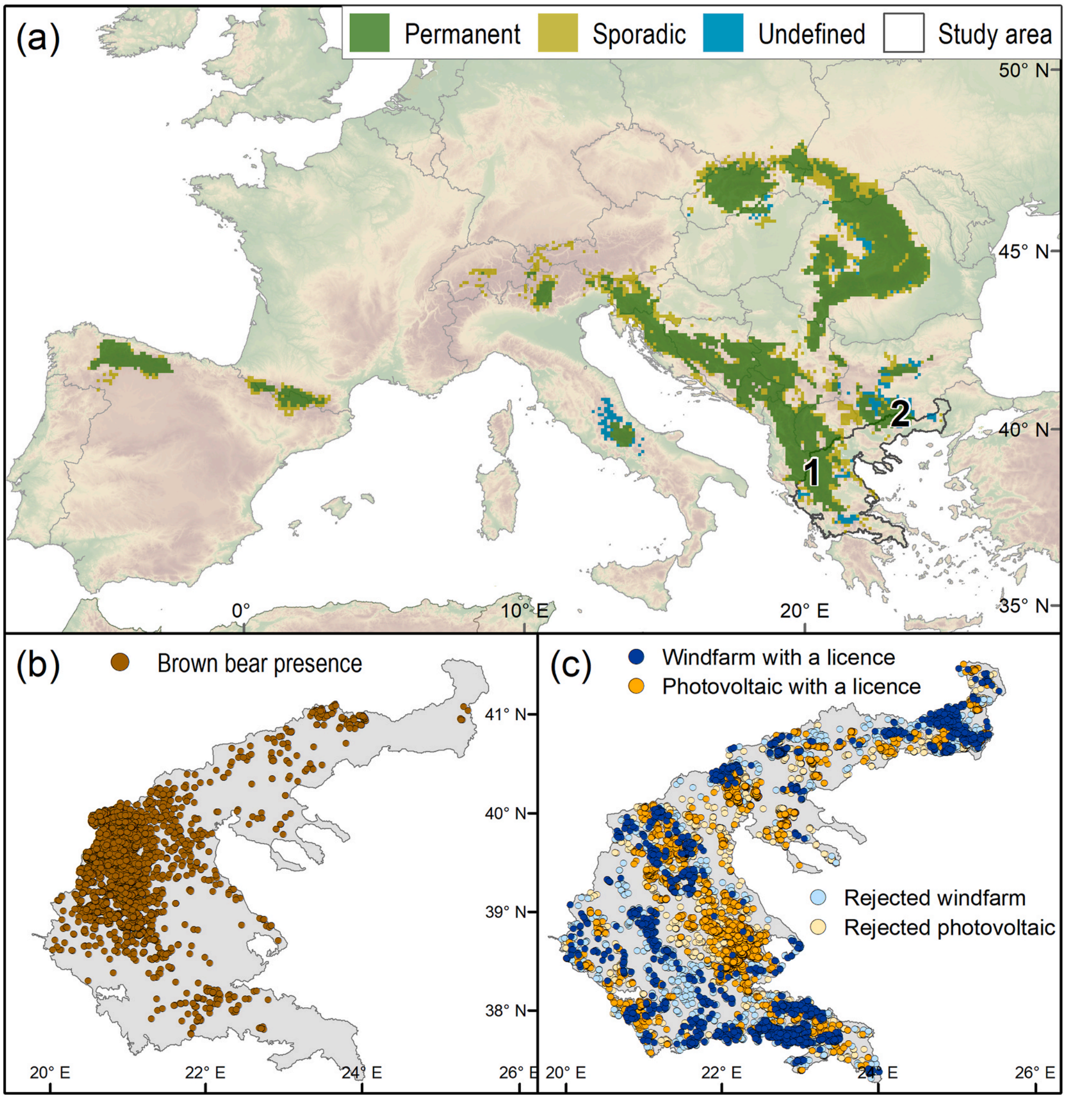


**Fig. 1.** (a) Location of the study area in Greece with respect to the distribution of the brown bear in Europe (10 × 10 km resolution), according to the Large Carnivore Initiative for Europe (Kaczensky et al., 2024). 1) Pindos Mountain Range, 2) Rhodope Mountain Range; (b) Spatial distribution of the brown bear presence-only dataset in Greece (2004–2024), presenting two main clusters of bear presence along the Pindos and Rhodope ranges, and sparse presence records in lowland basins in between; (c) Spatial distribution of Renewable Energy Source projects (i.e., windfarm and photovoltaic) in Greece (www.geo.rae.gr), as of December 2024, with windfarm projects concentrated mainly in mountainous areas (notably in the Pindos and parts of the Rhodope Mountain Range) and photovoltaic projects concentrated mainly in lowland/agricultural areas.

approach has clear advantages over other methods when using presence-only data (Renner et al., 2015) and is increasingly employed in species distribution modelling (Bourobou et al., 2023). We used our bear presence-only dataset as training data. We modelled the intensity of the PPM as a function of nineteen potential predictor variables (i.e., sixteen ecological variables influencing species presence and three observer bias variables affecting detection probability; Table 1). We included quadratic terms and pairwise interactions between ecological variables to capture non-linear relationships and interactions among all pairs of variables. All variables were standardised before modelling. Variable selection was performed using Least Absolute Shrinkage and Selection Operator (LASSO) regularisation (i.e., a regularisation path of 200 fitted models), which applies a penalty to regression coefficients, shrinking some to zero. In this way, it selects the most informative predictors while reducing overfitting, which is particularly valuable when using many correlated variables (Elith et al., 2011; Tibshirani, 1996). We used area-interaction models to account for spatial dependence among presence data, using an interaction radius of 2 km, which was the radius maximizing the pseudo-likelihood of the spatial point process for the species (See also Supplementary Fig. S1). Area-interaction models introduce an additional spatial term that captures clustering or dispersion among points, distinguishing ecological aggregation from artefacts of uneven sampling (Baddeley and van Lieshout, 1995). Point interactions might reflect a higher intensity in observation effort and were therefore considered as an additional observer bias variable.

Once the models, including both ecological and observer bias variables, had been fitted, we corrected for observer bias by setting all observer bias variables (i.e., including point interactions) to a standard value across the study area (in this case, the minimum value). In this way, we produced a final raster map with bias-corrected intensity estimates at a 1 $km^2$ resolution, which were rescaled to values ranging from 0 (lowest suitability) to 1 (highest suitability). This final map (hereafter referred to as the habitat suitability map) reflected the relative likelihood of presence, assuming uniform sampling effort throughout the region, and was considered a proxy for habitat suitability. As our habitat suitability model was based on presence-only data, which may be prone to observer bias, we used the most recent distribution map for the brown bear in Greece, produced by the LCIE (Kaczensky et al., 2024), as an

**Table 1**
Ecological and observer bias variables used as predictor variables for habitat suitability models based on inhomogeneous Point Process Models (PPMs).

| Variable | Unit | Description |
|---|---|---|
| **Ecological variables** | | |
| Elevation | m a.s.l. | Elevation above the sea level (a.s.l.)[a] |
| Roughness | Index | Terrain roughness index[a] |
| Broad-leaved forest | % | Percentage of broad-leaved forest[b] (CLC = 311) |
| Coniferous forest | % | Percentage of coniferous forest[b] (CLC = 312) |
| Mixed forest | % | Percentage of mixed forest[b] (CLC = 313) |
| Shrubland | % | Percentage of transitional woodland-shrub[b] (CLC = 324) |
| Moors and heathland | % | Percentage of moors, heathland and sclerophyllous vegetation[b] (CLC = 322–323) |
| Grasslands | % | Percentage of natural grasslands[b] (CLC = 321) |
| Pastures | % | Percentage of pastures[b] (CLC = 231) |
| Heterogeneous agricultural areas | % | Percentage of heterogeneous agricultural areas and permanent crops[b] (CLC = 241–244) |
| Open spaces | % | Percentage of open spaces with little or no vegetation[b] (CLC = 331–334) |
| Wetlands | % | Percentage of inland and maritime wetlands[b] (CLC = 411–423) |
| Artificial surfaces | % | Percentage of artificial surfaces[b] (CLC = 111–141) |
| Distance to arable land | m | Distance to non-irrigated and permanently irrigated arable land[b] (CLC = 322) |
| Distance to the forest | m | Distance to forest edge[b], calculated from the boundary of the forest layer (CLC = 311–313) using the Euclidean distance from cell centroids. |
| Distance to rivers | m | Distance to rivers[c] calculated from OSM hydrography using the Euclidean distance from cell centroids. |
| ***Observer bias variables*** | | |
| Distance to human settlements | m | Distance to cities, villages and residential areas[c], calculated from OSM populated places (city/town/village/hamlet) using the Euclidean distance from cell centroids. |
| Density of primary roads | km/km$^2$ | Density of motorway and primary roads[c], calculated as the total length per 1 km$^2$. |
| Density of secondary roads | km/km$^2$ | Density of secondary and tertiary roads[c], calculated as the total length per 1 km$^2$. |

[a] Derived from the Shuttle Radar Topography Mission (SRTM) 30 m digital elevation model (United States Geological Survey, 2006).
[b] Derived from CORINE Land Cover 2018 (100 m, seamless) (https://land.copernicus.eu/pan-euro-pean/corine-land-cover), aggregating the Level-3 land CORINE Land Cover (CLC) class for each variable indicated in the description. For each 1 km cell, we computed the % cover of each category.
[c] Derived from OpenStreetMap (OSM) data in layered GIS format (www.openstreetmap.org/#map=6/38.359/23.810).

independent dataset for model evaluation. This distribution map was available at a 10 × 10 km resolution; therefore, before evaluation, we upscaled our habitat suitability map by averaging values within each 10 × 10 km grid cell, and then evaluated accuracy and predictive performance using the Area Under the Curve (AUC) and True Skill Statistics (TSS). AUC quantifies the model's ability to discriminate between occupied and unoccupied cells, while TSS balances sensitivity and specificity, providing an evaluation of predictive success (Fielding and Bell, 1997).

To examine the relationship between RES development and bear habitat suitability, we extracted the suitability values from our habitat suitability map at the polygon centroids of each windfarm and photovoltaic project. Then, we repeated the process for an equal number of randomly generated points, which served as a baseline for comparison. To evaluate differences between random locations and RES projects, we used Beta regression modelling, as habitat suitability values were not normally distributed and ranged between 0 and 1 (R package "betareg"; Cribari-Neto and Zeileis, 2010). Beta regression is designed explicitly for bounded, proportional data and accommodates flexible variance structures and skewed distributions, making it particularly suitable for habitat suitability scores (Ferrari and Cribari-Neto, 2004). To assess whether RES projects with a licence were significantly more often located in areas with lower habitat suitability compared to the random expectation, we included the type of location (i.e., real vs. random) as a predictor variable. To assess whether rejected RES projects were located in areas with higher habitat suitability compared to projects with a licence, we included the type of project (i.e., with a licence vs. rejected) as a predictor variable.

### 2.3. Potential impact of RES development on bear habitat and distribution

We used the evaluation results obtained from the habitat suitability modelling to classify our bear habitat suitability map into polygons (i.e., reclassification and polygonisation operations) that delineated our study area as follows: 1) "Suitable habitat", where habitat suitability was above the threshold maximizing TSS (i.e., achieving an optimal balance between sensitivity and specificity, ensuring the best predictive capacity to classify true positives and true negatives correctly); 2) "Hotspot", where habitat suitability was above the threshold maximizing specificity (i.e., maximizing the proportion of true negatives correctly identified while minimizing false positives, thus, ensuring the maximum confidence level in identifying truly highly suitable habitat areas). These threshold criteria are widely recommended in presence-only modelling because they achieve an optimal balance between correctly classifying presences and absences, providing maximum certainty in identifying truly suitable areas (specificity) (Liu et al., 2013). Considering the seasonal home ranges of brown bears in Greece (De Gabriel Hernando et al., 2020), we excluded suitable habitat and hotspot polygons $<25$ km$^2$, as these were considered to be too small to be functional habitat units for the species' requirements. However, adjacent, smaller polygons that were spatially interconnected (i.e., $\leq 2$ km) and together formed an area $\geq 25$ km$^2$ were retained (i.e., buffer and dissolve operations), as they could function as "Suitable habitat" or "Hotspot" units.

Following, we overlapped (i.e., overlay operation) the "Suitable habitat" and "Hotspot" polygons with the distribution map of the LCIE (Kaczensky et al., 2024) to produce an updated distribution map of the brown bear in Greece and classified it as follows: 1) "Area of permanent bear presence" with "Suitable habitat"/"Hotspot", when they overlapped cells with a permanent bear presence (i.e., including records of reproduction or continuous presence); 2) "Recovery area" with "Suitable habitat"/"Hotspot", when they overlapped cells of non-permanent bear presence (i.e., including sporadic and undefined presence); 3) "Potential expansion area" with "Suitable habitat"/"Hotspot" when they overlapped areas where the presence of brown bears has not (yet) been recorded.

To evaluate the potential impacts of RES development on the "Suitable habitat" and "Hotspot" polygons, we calculated their spatial overlap (i.e., intersection operation) with ongoing RES projects (i.e., projects with a licence and projects under evaluation) at three spatial scales: 1) Facility scale: area covered by the polygons of the ongoing projects that accounted for the direct and indirect impacts, due mainly to land loss (e.g., logging, clearing vegetation, earthwork, etc.) and habitat exclusion (e.g., fencing, noise, frequent human presence); 2) Proximity scale: area within a 2 km radius around each ongoing project that accounted for the indirect impact on large carnivore habitat and/or behaviour (Ferrão da Costa et al., 2018), due to increased human accessibility, noise/visual disturbance, modification of prey communities, denning/reproduction inhibition, etc.; 3) Landscape scale: area within a 5 km radius around each ongoing project [i.e., based on the radius of the minimum brown bear home range (De Gabriel Hernando et al., 2020)] that accounted for the indirect impact on brown bear home ranges, due to changes in habitat use and movements. For each spatial scale, we calculated the percentage of the different "Suitable habitat"/"Hotspot" areas affected by the ongoing RES projects within the three different types of brown bear distribution in Greece (i.e., "Area of permanent bear presence", "Recovery" and "Potential expansion area").

Finally, we repeated the process to calculate the percentage of N2K areas within the "Suitable habitat" polygons overlapping ongoing RES projects.

## 3. Results

### 3.1. RES spatial development in relation to bear habitat suitability

Habitat suitability models demonstrated good predictive performance, as indicated by AUC (0.922) and TSS values (0.696). Habitat suitability was highest primarily in the Pindos and Rhodope Mountain Ranges (Fig. 2a), as well as in other smaller and isolated mountain ranges throughout the country. The density of windfarm projects was highest in northeastern and southern Greece, while the density of photovoltaic projects was highest in the central part of northern Greece and in Central Greece (Fig. 2b).

Beta regression modelling indicated that bear habitat suitability was significantly higher at windfarm projects compared to random points [Estimate = 0.316, SE = 0.018, $z = 17.31$, $p < 0.001$, 95 % CI (0.280, 0.352)]. For photovoltaic projects, habitat suitability was significantly lower [Estimate = −0.427, SE = 0.011, $z = -38.20$, $p < 0.001$, 95 % CI (−0.449, −0.406)] (Fig. 3a). Habitat suitability didn't show statistical differences between windfarm projects with a licence and rejected projects [Estimate = 0.007, SE = 0.025, $z = 0.28$, $p = 0.780$, 95 % CI (−0.042, 0.055)], while habitat suitability was significantly higher at photovoltaic projects with a licence than at rejected projects [Estimate = −0.045, SE = 0.014, $z = -3.30$, $p < 0.001$, 95 % CI (−0.071, −0.018)] (Fig. 3b).

### 3.2. Potential impact of RES development on bear habitat and distribution

We identified 29 disjunct habitat units in Greece with "Suitable habitat" and 37 disjunct units with "Hotspots" (Fig. 4a). "Suitable habitat" in "Areas of permanent bear presence" comprised two disjunct nuclei in the Pindos and Rhodope Mountain ranges in the western and eastern part of the country, while the same habitat comprised 15 and 12 disjunct areas in the "Recovery areas" and "Potential expansion areas" respectively. "Hotspots" comprised 15, 20 and two disjunct areas in the "Areas of permanent bear presence", the "Recovery areas" and the "Potential expansion areas", respectively. The potential impact of RES development on these areas is visualised in Fig. 4b and presented in Table 2.

At the facility scale, the spatial overlap between the "Suitable habitat" and ongoing RES projects was 2.6 %, with a higher proportion in the "Potential expansion areas" (4.4 %). At the proximity scale, this overlap increased to 26.8 %, with a higher overlap in the "Potential expansion areas". Finally, at the landscape scale, the spatial overlap increased further to 51.3 %, with a higher overlap again in the "Potential expansion areas" (Table 2). Windfarm projects contributed the most to these percentages at all scales (Fig. 5a). Similarly, the overlap at the facility scale of "Hotspot" areas with ongoing RES projects was 1.5 %, with a higher overlap in the "Recovery areas". At the proximity scale, this overlap increased to 17.6 %, with a higher overlap in the "Recovery areas". Finally, at the landscape scale, the overlap increased even further to 35.0 %, with a higher overlap, again, in the "Recovery areas" (Table 2). Ongoing windfarm projects in "Hotspots" contributed again the most to this overlap (Fig. 5b). Finally, the proportion of N2Ks within "Suitable habitat" for bears affected by ongoing RES projects was 67.5 %: the spatial overlap of these projects at the facility, proximity and landscape scale was 1.4 %, 14.8 % and 32.9 % respectively. In all three scales, the N2Ks mainly affected were within the "Potential expansion areas" (Table 2).

## 4. Discussion

Our findings contribute to a better understanding of the compatibility of current RES development in Greece with the national priorities for the conservation of biodiversity (e.g., EU, 2030 Biodiversity Strategy; see also Hermoso et al., 2022) and the Sustainable Development Goals of the Paris Agreement (Sachs et al., 2019), which seek to reduce greenhouse gas emissions while protecting and restoring biodiversity.

In the case of the spatial development of RES regarding bear habitat suitability in Greece, our results indicate that windfarms have/are being significantly more often developed in the most suitable bear habitat. This pattern was not observed in the case of photovoltaic projects, which were located in areas of lower habitat suitability. We speculate, however, that this was not due to a special consideration of the ecological requirements of bears, but moreover, because, in general, photovoltaic projects are developed in areas with open landscapes with low forest cover (Tinsley et al., 2024), which are less frequently used by brown bears (De Gabriel Hernando et al., 2021), and thus, identified by our models as less suitable. However, even such areas can play an essential role in the survival of the brown bear, as they can serve as feeding areas or as connectivity corridors, especially in the highly anthropogenic Mediterranean landscapes of southern Europe (Penteriani et al., 2020). Furthermore, it is also worthwhile to highlight the fact that rejected windfarm projects did not differ environmentally from accepted projects, indicating that other criteria have driven the rejection decision.

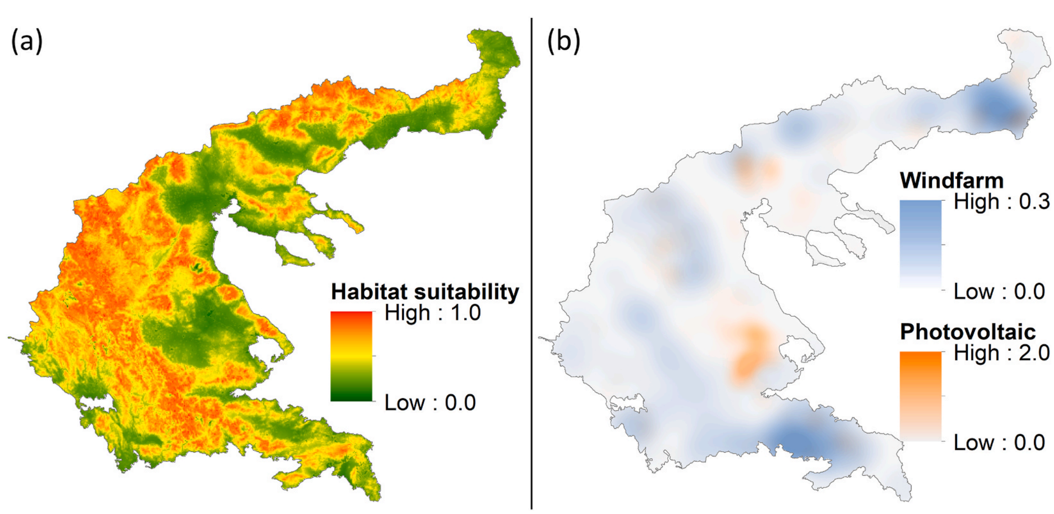


**Fig. 2.** (a) Map of the study area indicating habitat suitability for brown bears, as obtained through inhomogeneous Point Process Modelling. (b) Map of the study area indicating the Kernel density of windfarm and photovoltaic projects (N projects/km$^2$).

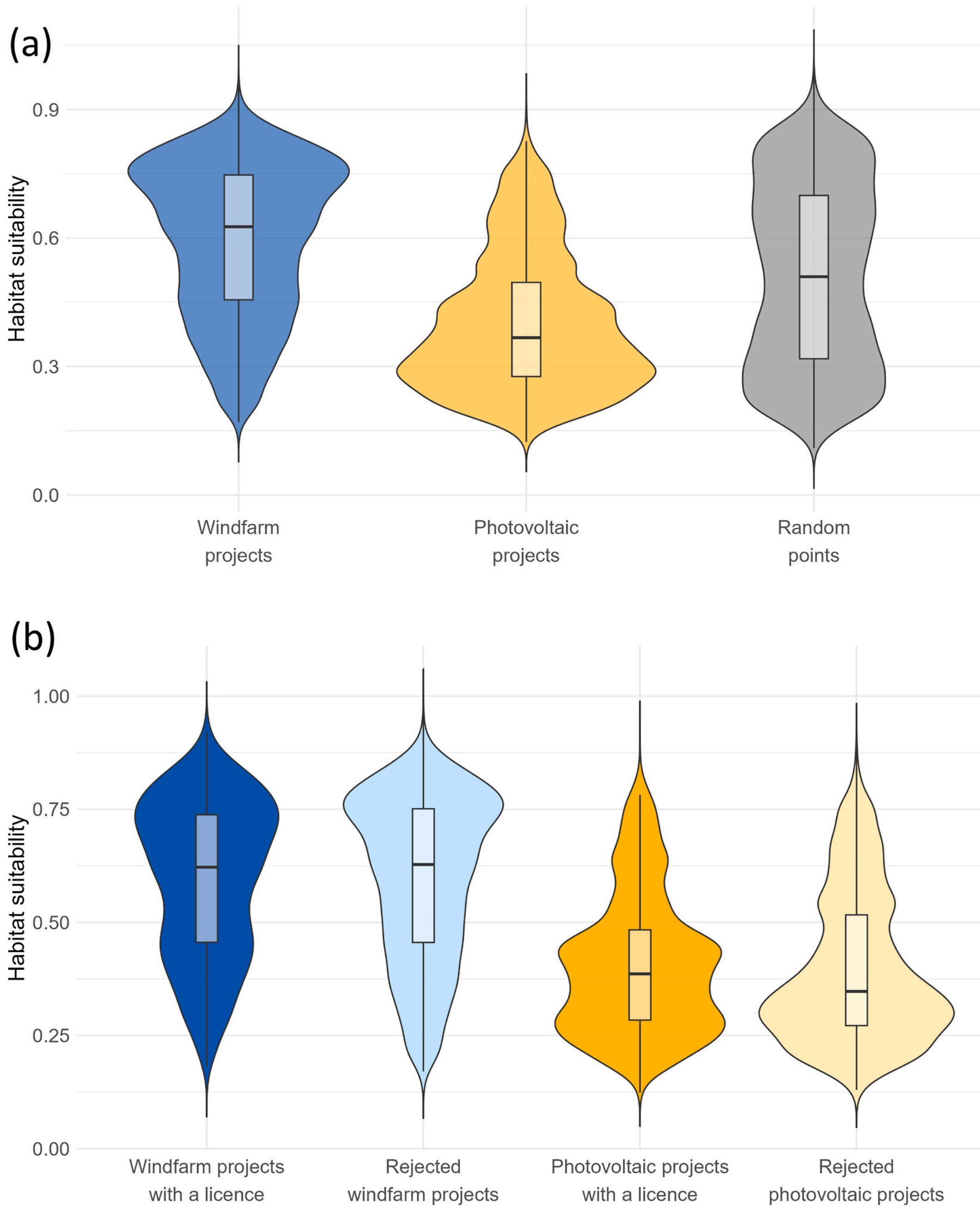


**Fig. 3.** (a) Comparison of brown bear habitat suitability between windfarm and photovoltaic projects and randomly selected points; (b) Comparison of brown bear habitat suitability between RES projects with a licence and rejected RES projects. The violin plots display the distribution of habitat suitability values for each group, with the inner boxplots representing the interquartile range (IQR) and the horizontal lines indicating the median.

Therefore, we claim that Greece has not paid the necessary attention to the spatial development of RES concerning a critical umbrella species and its habitat in the country. Considering the potential adverse effects of RES development, we firmly believe that risk assessments to biodiversity should be prioritised in the RES licensing process.

Unsound RES development has been linked to several adverse effects on biodiversity (Rehbein et al., 2020). If we suppose that the deficiencies in the current spatial development of RES remain, then bears in Greece will face a permanent habitat loss of 2.6 % of the "Suitable habitat" available to them, including 150 km$^2$ of "Hotspot" areas. Habitat loss has been identified as one of the most critical threats to brown bears in Europe (Swenson et al., 2020); however, it is particularly relevant in areas with a high anthropogenic footprint, such as those in southern Europe (Penteriani et al., 2020). In addition to habitat loss, at least half (i.e., 51.3 %) of the "Suitable habitat" available to the species in the country will be negatively affected by the increase in anthropogenic activity associated with the development of RES (i.e., during construction and operation). Anthropogenic disturbance affects bear behaviour and movement and is considered an essential threat to population connectivity and bear survival in Europe (Hertel et al., 2025). The areas most affected by this increase are the ones most closely related to the ongoing recovery of the species in the country (i.e., "Recovery" and "Potential expansion" areas); the current protection framework for bears and biodiversity in Greece does not appear to offer the appropriate safeguards against this increase in anthropogenic activity, as more than a third of N2Ks in the "Suitable habitat" are going to be negatively affected by RES. We conclude that the combined effects of habitat loss and increased anthropogenic activity caused by RES development, as it

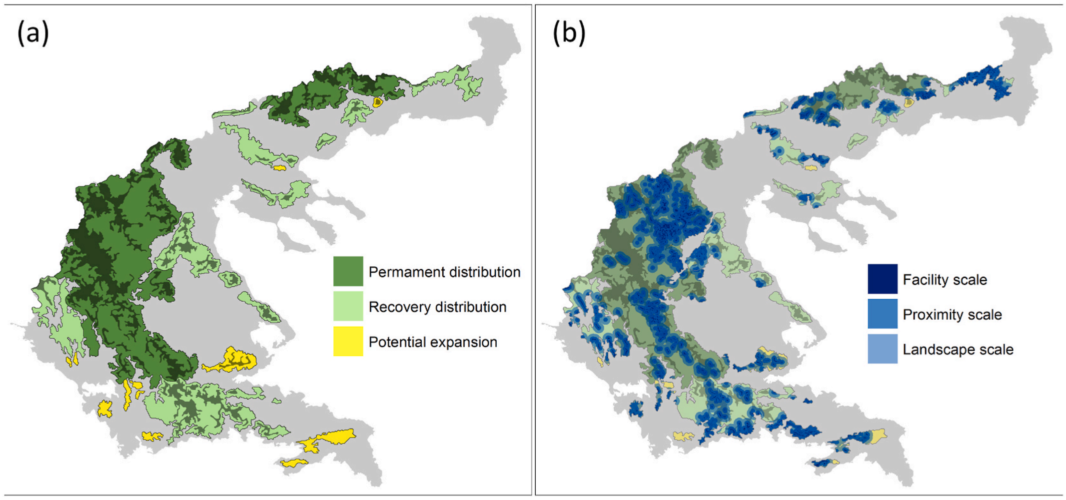


**Fig. 4.** (a) Map of the study area indicating brown bear habitat suitability (i.e., "Suitable habitat", outer boundary; "Hotspot", inner shaded area) and distribution in Greece, based on habitat suitability modelling and updated information on species presence in the country. "Suitable habitat" and "Hotspot" areas were derived from the 1-km suitability raster by reclassifying the values above the threshold maximizing TSS and the threshold maximizing specificity, respectively, and polygonised and filtered to retain functional units ≥25 $km^2$; (b) Map of the study area indicating the overlap between the brown bear habitat suitability and the distribution maps with the ongoing windfarm and photovoltaic projects in Greece, at the facility, proximity (i.e., 2 km buffer) and landscape (i.e., 5 km buffer) scales.

**Table 2**
Number of disjunct areas (N) with "Suitable habitat", "Hotspots" and Natura 2000 areas within the "Suitable habitat" of brown bears in Greece, affected by ongoing RES projects at three spatial scales (i.e., Facility, Proximity, Landscape).

| Distribution | Areas (N) | Areas affected | | Total area ($km^2$) | Facility scale (polygon) | | Proximity scale (2 km buffer) | | Landscape scale (5 km buffer) | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | N | % | | $km^2$ | % | $km^2$ | % | $km^2$ | % |
| **"Suitable habitat"** | | | | | | | | | | |
| Permanent | 2 | 2 | 100.0 | 22,649 | 537 | 2.4 | 5491 | 24.2 | 10,954 | 48.4 |
| Recovery | 15 | 13 | 86.7 | 12,976 | 344 | 2.7 | 3706 | 28.6 | 6899 | 53.2 |
| Potential expansion | 12 | 11 | 91.7 | 2142 | 95 | 4.4 | 935 | 43.7 | 1516 | 70.8 |
| Combined | 29 | 26 | 89.7 | 37,767 | 976 | 2.6 | 10,132 | 26.8 | 19,369 | 51.3 |
| **"Hotspot"** | | | | | | | | | | |
| Permanent | 15 | 13 | 86.7 | 7373 | 97 | 1.3 | 1153 | 15.6 | 2318 | 31.4 |
| Recovery | 20 | 17 | 85.0 | 2611 | 54 | 2.1 | 603 | 23.1 | 1163 | 44.5 |
| Potential expansion | 2 | 2 | 100.0 | 150 | 2 | 1.3 | 32 | 21.3 | 64 | 42.7 |
| Combined | 37 | 32 | 86.5 | 10,134 | 153 | 1.5 | 1788 | 17.6 | 3545 | 35.0 |
| **Natura 2000** | | | | | | | | | | |
| Permanent | 61 | 40 | 65.6 | 8316 | 109 | 1.3 | 1159 | 13.9 | 2625 | 31.6 |
| Recovery | 55 | 33 | 60.0 | 4170 | 51 | 1.2 | 594 | 14.2 | 1345 | 32.3 |
| Potential expansion | 12 | 8 | 66.7 | 686 | 30 | 4.4 | 200 | 29.2 | 366 | 53.4 |
| Combined | 114 | 77 | 67.5 % | 13,172 | 190 | 1.4 | 1953 | 14.8 | 4336 | 32.9 |

is currently being implemented, will harm the conservation prospects and ultimately the status of bears in Greece.

It should be highlighted that our study focuses only on the potential effects of RES development on land loss/land change (i.e., through vegetation removal, construction of the RES facilities) and the associated increase in anthropogenic pressure, without examining the impact of other related works, such as the development of disposal areas, the widening of the existing road network or the development of new access roads (Kati et al., 2023). The construction of new roads in suitable habitats may trigger a cascade of new impacts (e.g., fragmentation, further increase in anthropogenic activities, illegal activities; Selva et al., 2015), which may exacerbate the negative impacts on biodiversity even more. Our study also does not examine the cumulative effects of all these works, nor the synergistic effects with other human activities not related to RES development that are also expected to increase negative pressure on local biodiversity, e.g., the development of linear transportation infrastructure. Brown bears in Greece have recently been experiencing increasing pressure from habitat fragmentation due to the construction of major highways (Karamanlidis et al., 2012). Given the plethora of proven and potential, but realistic, negative impacts on the landscape, we conclude that the current spatial planning of RES development threatens the conservation prospects of bears in Greece, thus jeopardising the successful conservation efforts of the past 40 years. If we suppose that the current spatial development of RES is pursued, then Greece is unlikely to achieve its national goals for conserving its brown bear population, while jeopardising its efforts to meet international commitments related to biodiversity conservation, such as the EU 2030 Biodiversity Strategy. Furthermore, regarding the conservation prospects of the brown bear population in Greece, we believe that these are becoming increasingly negative. In addition to the adverse effects of RES development, wildfires in Greece have also taken a toll on the habitat available to the species in the country; wildfires have been increasing in Greece lately, disproportionately affecting protected areas. Wildfires are often associated with the operation of wind and solar energy facilities (Moustakas, 2025). Moreover, RES development in suitable bear habitat can contribute to another major threat to the species: human-bear conflicts. As RES infrastructures increase human permeability to bear core areas, the risk of negative human-bear interactions also increases, which is further exacerbated by habitat destruction, including wildfires (Blakey et al., 2022).

Given the results of our study and the intensive efforts that most countries in Europe (Kaczensky et al., 2024) and beyond have been

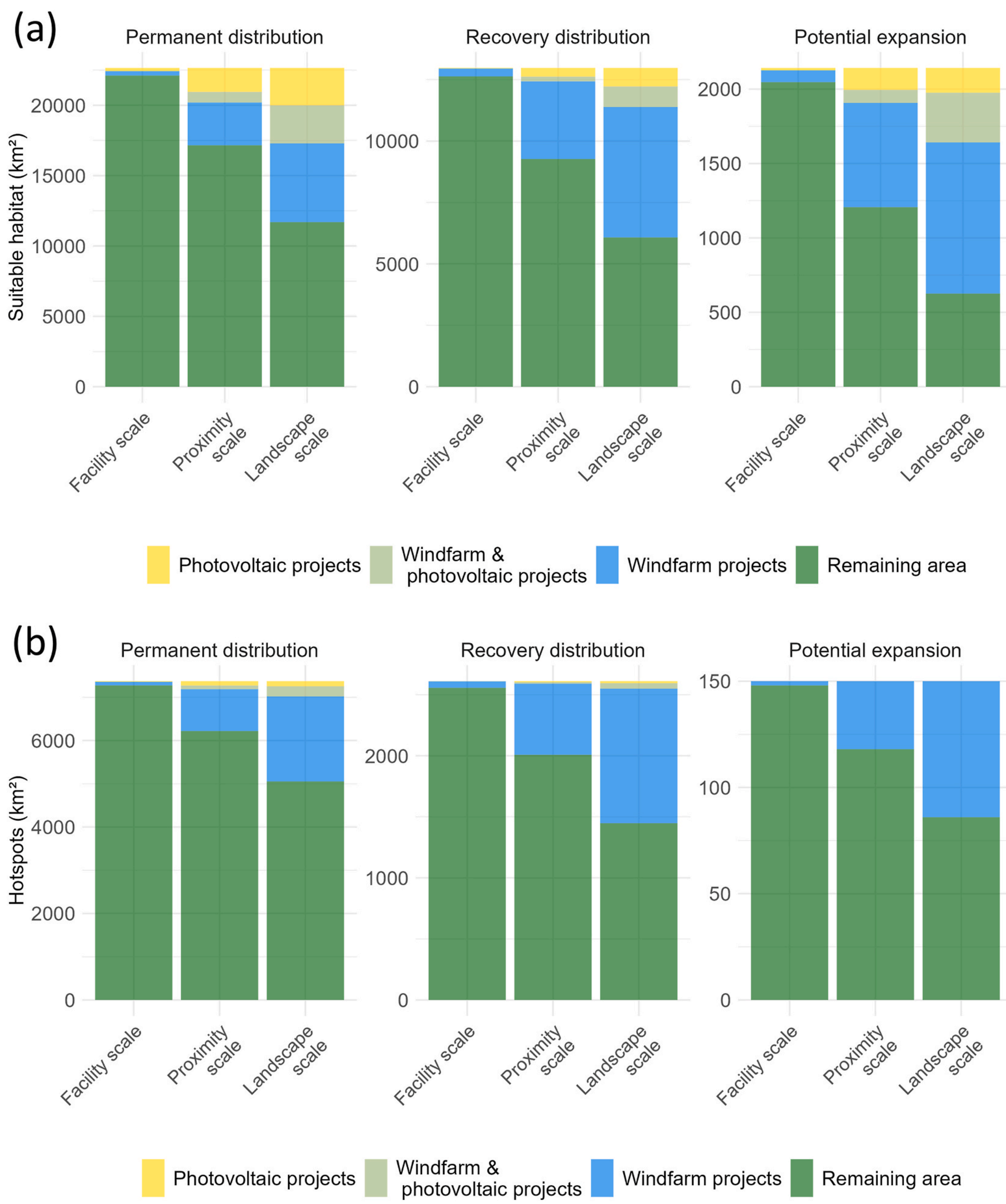


**Fig. 5.** Percentual representation of "Hotspots" (a) and "Suitable habitat" (b) for brown bears in Greece, according to the distribution category (i.e., Permanent, Recovery and Potential Expansion) and their overlap with ongoing RES projects at the facility, proximity and landscape scales.

investing in protecting their bear populations, we strongly urge European and national authorities in countries with bear populations to carefully assess whether their current spatial development of RES is compatible with the conservation priorities of these populations and other endangered wildlife.

### 4.1. Implications for conservation

This is the first study to assess the potential effects of RES development on brown bears; the adverse impact of RES development on the species is in line with the effects on other species in Greece (Moustakas et al., 2023) and worldwide (Rehbein et al., 2020). Considering the "Umbrella effect" of N2K in protecting biodiversity and the extent to which such areas are or will be affected by the current spatial development, we firmly believe that RES development will not only negatively affect bears but also biodiversity in Greece in general. This would align with findings from other parts of the world and reflect the emerging, concerning trend of RES development in areas with high biodiversity (Rehbein et al., 2020), including other biodiversity hotspots worldwide (e.g., Millon et al., 2018; Thaker et al., 2018). In the Mediterranean biogeographic region, adverse effects on biodiversity resulting from RES development have already been documented in birds in Spain (Serrano et al., 2020). If Greece were to follow in this direction, it would increase the negative pressure on this biodiversity hotspot. With an economy in transition (European Commission, 2024), Greece is in urgent need to include strategic, spatial planning in its biodiversity conservation planning process (Pereira and Cooper, 2006) and define science-based biodiversity conservation targets (Svancara et al., 2005) that will protect its unique biodiversity. Regarding the development of RES in Greece, we propose three priority actions: 1) Spatial planning of RES development in Greece should go back to the drawing board and take into account the findings of the current study and others about the conservation of biodiversity in the country (e.g., Kati et al., 2021; Moustakas et al., 2023) to define appropriate areas for RES investment that will not conflict with priority areas for biodiversity conservation. Inclusion of biodiversity-relevant aspects should become standard in the multi-criteria decision process of RES licensing, as is the case for other socio-economic, cultural and environmental issues (Mourmouris and Potolias, 2013); 2) Strengthening quality controls of the necessary assessments to approve the development of RES projects in Greece (Vasilakis et al., 2017) to account for the ecological requirements of brown bears, and other wildlife; and 3) Systematic monitoring of the effects of ongoing RES projects on bears and other conservation priority species that will contribute to an adaptive management that minimises the impacts of RES on biodiversity. Incorporating remote-sensing monitoring, following the approaches used to detect long-term habitat change (e.g., Valjarević et al., 2018) would enable before–and–after assessments of habitat conversion attributable to RES.

In conclusion, we provide concrete evidence that, despite the existence of a regulatory framework (i.e., often lacking or weaker in other countries with high biodiversity), current RES development in this biodiversity hotspot will negatively affect biodiversity, suggesting that current planning regulations are not being adhered to and are not fully effective. Given the continuous decline in biodiversity and the rapid development of RES worldwide (Rehbein et al., 2020) we strongly urge all stakeholders to proceed with "adaptive planning" regarding the further development of RES (Kato and Ahern, 2008), where not only concerns about the effects on biodiversity are addressed, but also other (future) uncertainties, such as changes in climate and economic policy, are considered (Tinsley et al., 2024).

## CRediT authorship contribution statement

**Alexandros A. Karamanlidis:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Miguel de Gabriel Hernando:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Methodology, Formal analysis, Data curation, Conceptualization.

## Declaration of competing interest

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.

## Acknowledgments

The study is dedicated to Kir Yiannis Boutaris and his unwavering spirit to swim against the tide. It was carried out within the framework of the project "Large Carnivore Connect: targeted conservation actions for maintaining/promoting large carnivore connectivity in the south-western Balkans", supported by the Prespa Ohrid Nature Trust. PONT support for the production of this publication does not constitute an endorsement of the contents which reflects the views only of the authors, and PONT cannot be held responsible for any use which may be made of the information contained therein.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2025.127794.

## Data availability

Data will be made available on request.

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