The wind of change: Mapping wind energy growth and multi-species vulnerability in the Mediterranean

Chiara Costantino*[1], Jacopo Cerri[1,2], Ilaria Fozzi[1], Davide De Rosa[1], Fiammetta Berlinguer[1]

[1]Department of Veterinary Medicine, University of Sassari, Via Vienna 2, 07100, Sassari, Italy

[2]Mammal Research Institute, Polish Academy of Sciences, Stoczek 1, Białowieża, 17-230, Poland

Abstract

The rapid expansion of wind energy across the Mediterranean region calls for more advanced tools to assess and mitigate its impacts on biodiversity.
In this study, we propose an innovative approach that integrates historical satellite imagery and ecological modelling to assess the spatiotemporal overlap between wind energy development and habitat suitability for multiple vulnerable raptor species.
We reconstructed a 13-year trajectory of wind turbine distribution using high-resolution satellite images and applied species distribution models (SDMs) to eight raptor species of conservation concern. Our analysis revealed a marked increase in wind energy infrastructures, with a high degree of overlap between newly developed areas and suitable habitats for multiple species.
This approach highlights the potential of combining geospatial data, predictive modelling, and a multi-species perspective to complement traditional assessment methods.
Our results offer useful insights for identifying priority areas for monitoring and mitigation and suggest a transferable framework that could support more biodiversity-informed energy planning in Mediterranean ecosystems.

Keywords: multi-species assessment, raptors, bird conservation, satellite imagery, remote sensing, renewable planning

This is a preprint version of a study currently under consideration for publication in a peer-reviewed journal.

---

*Corresponding author: c.costantino@studenti.uniss.it

1 Introduction

Over the past decade, wind energy has grown rapidly, establishing itself as a key source of renewable energy and reaching a global capacity of 651 GW in 2019 (Global Wind Energy Council, 2020). This positive trend, reinforced by the agreement reached at the 2015 United Nations Climate Change Conference (COP 21) in Paris, underscores the crucial role of renewable energy in the global energy transition, with wind power standing out as one of the key contributors.

However, despite the environmental benefits of reducing greenhouse gas emissions, wind turbines can negatively impact wildlife, particularly flying vertebrates such as birds and bats (Marques et al., 2014; Jonasson et al., 2024). Impacts include direct mortality from collisions with rotor blades and power lines (Rydell et al., 2010; Garvin et al., 2011; Huso et al., 2016) and indirect impacts such as habitat loss and disturbance, which can lead to species displacement and reduced population viability (Drewitt and Langston, 2006; Madders and Whitfield, 2006; Zimmerling et al., 2013; Arnett and May, 2016; Frick et al., 2017).

Moreover, wind-power plants may act as barriers to movement, with the severity of these effects varying depending on site-specific and species-specific factors. However, birds can exhibit behavioural adaptations to mitigate these impacts, such as fleeing, shifting activity patterns, or modifying habitat utilization; these responses are collectively termed avoidance (May, 2015). Among avian species, those that use soaring flight, such as raptors, are particularly vulnerable to the impacts of wind farms.

The latter, in fact, cause high mortality among them due to collisions with turbine blades, recorded in various areas (Hunt, 2002; Barrios and Rodríguez, 2004; de Lucas et al., 2008). Their vulnerability can be attributed to three principal factors: low reproductive rates and delayed sexual maturity, which make populations incapable to compensate for additional mortality (Duriez et al., 2023); limited visual capacity in the direction of movement, impairing their ability to detect vertical obstacles such as turbine blades (Martin et al., 2012); and the overlap between wind farm locations and geomorphological features, such as ridges and updrafts, which are attractive for raptor activities (Katzner et al., 2012; Poessel et al., 2018; Rushworth and Krüger, 2014). Other factors may also influence collision risk. For example, species with high wing loading and weak-powered flight, like Griffon Vultures (*Gyps fulvus*), rely on uplift winds to stay aloft. In poor uplift conditions, they are forced to fly lower, increasing their risk of collision with turbine blades (De Lucas et al, 2008; Pennycuick 1975, 1998; Janss and Ferrer, 2000).

Moreover, species that habitually fly at dawn, dusk or during the night may have a lower ability to detect and avoid turbines (Larsen and Clausen, 2002). In Europe, the precautionary principle is widely adopted to mitigate the impacts of wind turbines on threatened species (Braunisch et al., 2015; Kriebel et al., 2001). One of the most common applications of this principle is the buffer zone approach, which excludes wind turbines from areas around sensitive locations such as nesting sites. Buffers are typically defined based on expert knowledge or the estimated home range size of sensitive species (Bright et al., 2008; Janss et al., 2010; Venter et al., 2019).

This method is relatively easy to implement and avoids the uncertainties inherent in complex risk models, which often rely on multiple parameters with unknown distributions or inaccurate spatial data. However, it has notable limitations. Buffer zones are typically defined based on habitat use during a single life stage, most commonly the reproductive phase, without accounting for the spatial and temporal variability of species movements and habitat utilization.

Furthermore, as a static approach, buffer zones lack predictive capacity, making them insufficient for anticipating future conflicts, especially in dynamic systems where populations are expanding, declining, or shifting their distributions (Hirzel et al., 2004; Krüeger et al., 2014; Braunisch et al., 2015). Rather than abandoning buffer zones entirely, a more effective strategy would be to enhance them by integrating predictive tools. This combination would allow buffer zones to account for ecological dynamics, improving their adaptability to changes in species distribution and movement patterns. As a result, conservation measures would be not only scientifically robust but also more practical and effective in the long term.

To fill this gap, this study provides a novel spatiotemporal assessment of wind energy expansion over the past 13 years and its impact on the vulnerability of eight species of raptors of conservation concern. Unlike previous studies, we reconstruct the spatial and temporal patterns of wind farm development using repeated historical aerial imagery. This approach allows us to track changes in turbine distribution over time and identify areas where cumulative impacts are likely to be most significant. Despite the recognized importance of cumulative impacts (Willsteed et al., 2018), spatiotemporal analyses of wind turbine expansion remain largely unexplored in the literature.

Moreover, while most studies focus on single species, our work assesses the effects of wind energy on multiple vulnerable species simultaneously. This broader perspective is crucial, as wind energy impacts extend beyond direct mortality and can also alter ecological interactions between species, such as predation and competition (Sergio and Hiraldo, 2008). By integrating Species Distribution Models (SDMs), we identify high-risk areas where these effects may be particularly relevant, providing a more comprehensive understanding of the ecological consequences of wind energy.

Our case study in Sardinia, Italy, exemplifies this approach by combining spatiotemporal wind energy mapping with a multi-species assessment. While Sardinia's wind energy expansion has specific local characteristics, the methodology we adopt provides a transferable framework applicable to other regions. These insights contribute to more effective conservation planning and the sustainable development of renewable energy.

2 Methods

2.1 Study area

Sardinia, the second-largest island in the Mediterranean Sea, spans 24,094 km^2 and hosts a diverse range of Mediterranean habitats of conservation concern. These ecosystems are safeguarded by a network of Natura 2000 sites covering 8,646 km^2 and regional and national parks extending over 1,273 km^2.

In recent years, the island has experienced a significant expansion of wind energy, with a steady increase in installed capacity (Cerri et al., 2024a). While playing a strategic role in renewable energy development, Sardinia's landscapes also reflect a long-standing pastoral tradition that has shaped local ecosystems and contributed to biodiversity conservation.

The island is home to several species of high conservation value, including the Griffon Vulture (*Gyps fulvus*), currently the focus of the LIFE Safe for Vultures project, which aims to support its protection and population recovery (https://lifesafeforvultures.eu/). Other vulnerable or endemic species include the Golden Eagle (*Aquila chrysaetos*) and the Red Kite (*Milvus milvus*). Sardinia also harbours Italy's last remaining population of the Little Bustard (*Tetrax tetrax*), along with two endemic species of conservation concern: the Sardinian Long-eared Bat (*Plecotus sardus*) and the subspecies Sardinian Goshawk (*Accipiter gentilis arrigonii*) .

Other vulnerable raptors, such as Bonelli's Eagle (*Aquila fasciata*), are being reintroduced on the island through the EU-funded LIFE ABILAS project (LIFE23-NAT-IT, 2024–2030). With growing pressure from wind energy infrastructure and the presence of multiple vulnerable species, Sardinia represents a key area for assessing the balance between renewable energy expansion and biodiversity conservation.

2.2 Temporal and spatial distribution of wind turbines

This study incorporates the dataset developed by Cerri et al. (2024a), updated to 2023, which served as the reference for analysing temporal and spatial variations in the distribution of wind turbines in Sardinia. The dataset was constructed by comparing three pre-existing datasets and validating them through satellite imagery, with the latest update conducted using Google Earth

(https://www.google.it/intl/it/earth/index.html). A grid of 1 km^2 cells was employed, and each cell was checked against historical satellite images to ensure consistency within a 5 km buffer around each turbine.

To complement this, satellite images from 2011, 2014, 2017, and 2020 were used for a manual count of turbines visible in each time interval (Fig. 1). These years were chosen because 2020 marked a peak in applications for new wind farm projects in Sardinia, while the preceding years showed a gradual increase in installations.The selection of these specific time windows also provided a random temporal sampling framework.

The dataset primarily includes the geographic position of turbines, as identified through satellite imagery. The analysis of historical images allowed us to trace the evolution of wind energy infrastructure back to 2011, starting from the 2023 dataset and working retrospectively. These results provide a detailed spatial and temporal perspective on wind energy development in Sardinia, offering insights into the factors driving turbine placement and their implications for land-use changes.

To identify the spatiotemporal patterns of wind energy expansion on the island, we compared normalised wind turbine density rasters from 2011 and 2023. Raster cells were classified into two categories. Historically active areas include cells where wind energy activity was already present in 2011 and remained active in 2023 (value > 0 in 2011). As wind farms are a permanent alteration of the landscape, once wind turbines are built, these areas of the island are arguably those where sensitive species experienced the highest cumulative impacts.

Expansion zones include cells where wind energy activity was present in 2023 (value > 0) but absent in 2011 (value = 0), representing new wind farm developments established over the past decade.

2.3 Data selection

For this study, we selected eight raptor species based on specific criteria established for their inclusion in the species distribution models (SDMs). These criteria ensured that only species with sufficient occurrence data and ecological relevance for the study area were included in the models.

The species analysed using SDMs were: Golden Eagle (*Aquila chrysaetos*), (*Athene noctua*), Western Marsh Harrier (*Circus aeruginosus*), Peregrine Falcon (*Falco peregrinus*), Common Kestrel (*Falco tinnunculus*), Red Kite (*Milvus milvus*), and Common Buzzard (*Buteo buteo*). Table 1 provides an overview of the IUCN status at different levels and key information on the habitat preferences of these species.

An exception was made for the Griffon Vulture (*Gyps fulvus*), as its population has been supplemented through restocking efforts. Due to its status, this species was not included in the SDM analysis but was still considered in this study (see below).

Species were selected based on specific inclusion criteria. Only those with recognized conservation importance were considered, with priority given to species listed in Annex I of the Birds Directive or included in international conservation agreements. All selected species are indeed protected under the Birds Directive (2009/147/EC), with several listed in Annex I, which mandates the establishment of Special Protection Areas (SPAs) to safeguard their populations.

Additionally, many of these species are included in the annexes of key international agreements, such as the Bern Convention (Annex II) and the Bonn Convention, highlighting their conservation priority at both European and global scales. Additionally, species were included only if sufficient, high-quality occurrence data were available, ensuring the construction of statistically robust and ecologically meaningful species distribution models (SDMs, Stockwell and Peterson, 2002).

Ecological relevance was also a key factor, with priority given to species exhibiting flight behaviours or habitat preferences known to interact with wind farm infrastructure (e.g., De Lucas et al., 2008), allowing for a more effective assessment of potential impacts. The selected raptor species exhibit varying degrees of vulnerability to anthropogenic pressures, differing in their ecology, flight behaviour, and habitat specialization. This variability allows for a comprehensive assessment of how wind farm installations might affect species with different ecological niches and sensitivities.

Notably, the Bonelli's Eagle (*Aquila fasciata*) was not included in the dataset. This exclusion is due to the temporal scope of our analyses, which date back to 2011, when the species had not yet been reintroduced to the island. The reintroduction project, known as Aquila a-Life, commenced between 2018 and 2023, during which 39 Bonelli's Eagles were released in Sardinia (Raganella Pelliccioni et al., 2024).

Species for which limited data were available, such as Eleonora's Falcon (*Falco eleonorae*) and the Little Bustard (*Tetrax tetrax*), were also excluded. The scarcity of observations and reliable ecological records for these species prevented the development of robust models, which could have led to biased or unrepresentative outcomes.

2.4 Species distribution models

The models were constructed using four algorithms commonly used in SDMs studies, using the package Biomod2 (ftp://137.208.57.37/pub/R/web/packages/biomod2/biomod2.pdf) in RStudio: Random Forest (RF), Generalized Linear Models (GLM), Generalized Boosting Models (GBM) and Generalized Additive Models (GAM). These algorithms represent a mix of conventional statistical approaches and machine learning techniques, widely adopted for their ability to capture complex relationships between environmental variables and species distribution (Guisan et al., 2017). To ensure a robust spatial representation, models were developed at a 1 km resolution (Seo et al., 2008), providing a suitable balance between ecological relevance and computational efficiency in predicting species-habitat relationships.

Species presence data were obtained through a data-sharing agreement with Ornitho

(https://www.ornitho.it/), ensuring access to high-quality occurrence records for the study area.

From the models generated using the four algorithms, the best-performing model for each species was selected based on the best-performing model. The best model was selected using an aggregated score derived from sensitivity and specificity metrics. Subsequently, the True Skill Statistic (TSS) and Receiver Operating Characteristic (ROC) scores were calculated (Segurado Araújo, 2004; Elith et al., 2006; Prasad et al., 2006).

For each species, 10,000 pseudo-absences were generated, a choice supported by Barbet-Massin et al. (2012), who highlight how this number maximizes accuracy in regression models (GLM, GAM) by better representing unfavourable conditions. In ensemble models, the adoption of a unified set of pseudo-absences ensures consistency across algorithms, and a large number such as 10,000 serves as an effective compromise.

To further minimize spatial biases in presence data, the *spThin* package in R was used to perform spatial thinning, ensuring independent sampling by removing occurrence records violating a specified minimum nearest-neighbour distance (Aiello-Lammens et al., 2015). Additionally, to ensure that the model accounted for a realistic accessibility margin for the species, we applied a 5 km buffer around occurrence points. This choice is based on the concept of the accessible area described by Barve et al. (2011), which emphasizes that the spatial extent of the model should reflect the species' ability to explore the landscape over time, thereby reducing the risk of bias in model outcomes.

All environmental variables were standardised and centred at a spatial resolution of 1 km^2. The selection of environmental variables (shown in Table 2) was further refined by means of a Variance Inflation Factor (VIF) analysis, using usdm package in RStudio with an exclusion threshold of 5, in line with the study conducted by Shrestha in 2020, aim to exclude multicollinearity among the predictors in the regression model.

A threshold value of 0.50 was selected to consider only areas with habitat suitability values equal to or greater than 0.50, excluding lower suitability areas from the analysis. This threshold is commonly used in species distribution models to distinguish suitable from unsuitable habitats (Manel et al. 1999; Bailey et al, 2002; Woolf et al., 2002) The species considered in the analysis, together with ROC, TSS and other relevant information, are shown in Table 1.

rotating

Finally, a cumulative habitat suitability map (Fig. 4) was created by summing all individual species' raster layers. This approach allowed for the identification of areas with overlapping high suitability for multiple raptor species. The resulting map highlights zones that are not only suitable for several species but also those most impacted by wind energy development.

Species	**Median TSS**	**Median ROC (AUC)**	**Habitat**	**Italian IUCN status**	**Global IUCN status**	**Sardinian IUCN status**
Golden Eagle	0.75	0.94	Mountains, hills, grasslands, cliffs and open areas.	Near Threatened (NT)	Least Concern (LC)	Vulnerable (VU)
Little Owl	0.68	0.94	Agricultural areas, pastures, forest edges. Close to human's settlement.	Least Concern (LC)	Least Concern (LC)	Least Concern (LC)
Western Marsh Harrier	0.76	0.95	Wetlands, marshes, lakes, extensive reed beds.	Vulnerable (VU)	Least Concern (LC)	Near Threatened (NT)
Peregrine Falcon	0.70	0.91	Cliffs, mountains and urban areas.	Least Concern (LC)	Least Concern (LC)	Near Threatened (NT)
Common Kestrel	0.80	0.96	Open, urban, and rocky areas.	Least Concern (LC)	Least Concern (LC)	Least Concern (LC)
Red Kite	0.75	0.94	Semi-open areas with forests, agricultural areas, pastures and hillsides.	Vulnerable (VU)	Least Concern (LC)	Critically Endangered (CR)
Common Buzzard	0.80	0.97	Open woodland, grassland, hills and farmland.	Least Concern (LC)	Least Concern (LC)	Least Concern (LC)

Table 1: Median values of the True Skill Statistic (TSS) and area under the ROC curve (AUC) for species included in SDMs. The predominant habitat and conservation status according to the IUCN classification at the Sardinian (Shenk, 2009), national (Italy), and global levels are also indicated.

Variable	Description	Source	Reference
BIO5	Maximum temperature of the warmest month	CHELSA	`https://chelsa-climate.org/downloads/`
BIO12	Annual precipitation	CHELSA	`https://chelsa-climate.org/downloads/`
BIO18	Precipitation of the warmest quarter	CHELSA	`https://chelsa-climate.org/downloads/`
DEM	Digital Elevation Model	CGIARCSI	`https://csidotinfo.wordpress.com/data/srtm-90m-digital-elevation-database-v4-1/`
Distance to lakes	Euclidean distance from lakes	Hydro SHEDS	`https://www.hydrosheds.org/products/hydrolakes`
Slope	Terrain slope derived from DEM	CGIARCSI	`https://csidotinfo.wordpress.com/data/srtm-90m-digital-elevation-database-v4-1/`
Aspect	Terrain exposure derived from DEM	CGIARCSI	`https://csidotinfo.wordpress.com/data/srtm-90m-digital-elevation-database-v4-1/`
GHS	Global human settlement density	Copernicus JRC Data Catalogue	`https://human-settlement.emergency.copernicus.eu/download.php`
Crops and Pastures	Land cover classification (source pending)	LP DAAC	`https://developers.google.com/earth-engine/datasets/catalog/MODIS_006_MOD44B?hl=es-419`
Tree Percentage Cover	Percentage of tree coverage	LP DAAC	`https://developers.google.com/earth-engine/datasets/catalog/MODIS_006_MOD44B`
Wetlands	200m buffer around rivers	HydroSHEDS	`https://www.hydrosheds.org/products/hydrorivers`

Table 2: Description of the environmental datasets used for species distribution models (SDMs).

2.5 Griffon Vulture range delineation

Unlike other species analysed using SDMs approach, the spatial distribution of the Griffon Vulture was defined using GPS telemetry, in view of its history of restocking and active management. The current population derives from repopulation programs carried out in two key areas of Sardinia: the northwest (Bosa area) and the southeast (Villasalto area).

Between 2016 and 2021, the LIFE Under Griffon Wings project

(LIFE14 NAT/IT/000484, http://www.lifeundergriffonwings.eu/it/index.html) was implemented to improve the conservation status of the species, with a restocking program that included the release of 64 individuals in northwestern Sardinia.

To date, the species is currently part of the LIFE Safe for Vultures project (LIFE19 NAT/IT/000732, https://www.lifesafeforvultures.eu/), which aims at securing population growth and at enlarging its distribution. The delineation of the species' range was based on known colony location data, which allows to define the spatial distribution.

The area of presence was defined using a Dynamic Brownian Bridge Movement Model (dBBMM, Kranstauber et al., 2012), which estimates the cumulated occurrence distribution. The 95% isopleth was used as a presence/absence cutoff, defining the core activity range of the species. This threshold includes 95% of the estimated movement, effectively delineating areas of regular use while excluding exploratory and occasional movements (Fig. 5).

3 Results

In 2011, there were 529 wind turbines installed across Sardinia. Over the following 13 years, this number more than doubled, reaching 1,155 turbines by 2023. The growth was uneven, with distinct phases of more intense development. Between 2011 and 2014, the number of turbines increased moderately, rising to 624 turbines.

A significant expansion occurred between 2014 and 2017, with the number of turbines growing to 952, marking the most substantial increase within the study period. From 2017 to 2020, the construction of new turbines continued at a slower pace, reaching 1,119 turbines. Finally, between 2020 and 2023, turbines increased up to 1,155 units (Fig. 1).

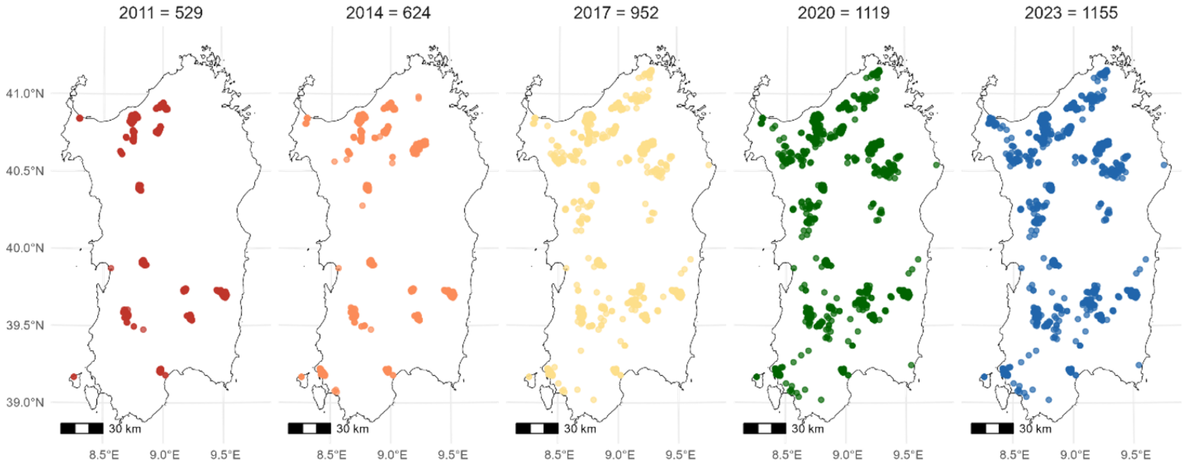

Figure 1: Spatial distribution of wind turbines installed in Sardinia between 2011 and 2023, grouped by year of installation. Each panel shows the cumulative number and geographic spread of turbines for a given year, highlighting the progressive expansion of wind energy infrastructure over time.

Historically active areas (Fig. 4a) are primarily distributed across the north-western part of the island, with additional concentrations in central and southern sectors. Their spatial pattern tends to form contiguous clusters of varying extent.

Expansion zones (Fig. 4b) appear more fragmented in comparison with historically active areas, affecting different and partly new portions of the island, including eastern, southern, and central sectors. In some cases, the new cells are located near areas already active in 2011, while in others, they occur as isolated cells, distant from the main previous developments.

Species Distribution Models (SDMs) developed for the seven raptor species showed a good level of predictive accuracy. Performance metrics, including the Area Under the Curve (AUC) and the True Skill Statistic (TSS), are summarized in Table 1. All models achieved AUC values above 0,90, indicating strong model discrimination capabilities. TSS values ranged between 0,68 and 0,80, further supporting the robustness of the models in distinguishing suitable from unsuitable habitats.

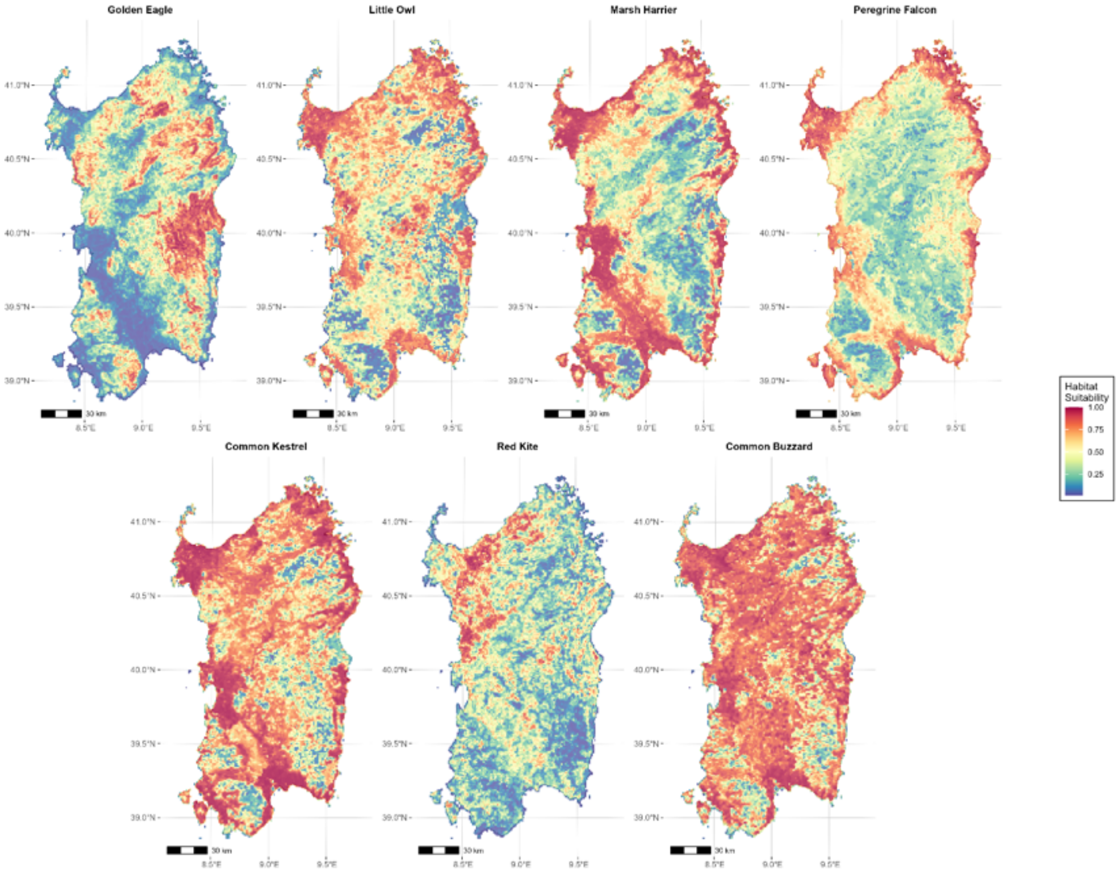

Figure 2: Habitat suitability maps for seven raptor species in Sardinia based on Species Distribution Models (SDMs). Each panel represents the spatial distribution of habitat suitability for a single species, with values ranging from low (blue) to high (red).

The habitat suitability maps, shown in Fig. 2, highlight areas across Sardinia with varying degrees of suitability for each of the seven raptor species. Among these species, the Golden Eagle shows a preference for mountainous and hilly areas, particularly in central and northern Sardinia. High-suitability habitats are mainly concentrated in elevated areas, where the species finds optimal nesting and foraging conditions. By 2011, 274 turbines (51.80% of the total at that time) were already located in areas suitable for the Golden Eagle, and by 2023, this number had risen to 443 turbines.

In contrast, the Little Owl demonstrates a widespread distribution, with high suitability areas concentrated in lowland agricultural landscapes, especially in central and western Sardinia. In its suitable areas, there were 337 turbines in 2011, and by 2023, this number had increased to 803 turbines (69.52% of those in the latest 2023 dataset).

Despite occupying different habitats, the number of turbines in areas suitable for the Marsh Harrier has also increased. In fact, the Marsh Harrier's suitable habitats are mainly concentrated along the coastal wetlands and lowland areas, particularly in the western and southern parts of Sardinia.

While in 2011 there were 148 turbines in its suitable areas, in 2023 this number had increased to 456 turbines. Similarly, the Peregrine Falcon shows high habitat suitability in rugged coastal cliffs and inland rocky outcrops, especially along the western coast and in parts of central Sardinia. There were 153 turbines in its suitable areas in 2011, and by 2023, this number had increased to 368 turbines (31.86% of the total turbines in 2023).

Red Kite shows a preference for open lowlands and hilly regions, particularly in central and northern Sardinia. High suitability areas are often associated with agricultural lands interspersed with woodlands, which provide optimal foraging conditions. The number of turbines in its suitable areas increased from 169 in 2011 to 383 in 2023, making up 33.16% of the total turbines installed in 2023.

The Common Kestrel and the Common Buzzard exhibit some of the broadest distributions, with high suitability spread across much of Sardinia. They thrive in open agricultural landscapes, grasslands, and mixed habitats, leading to extensive areas of moderate to high suitability across the island. As a result, in both 2011 and 2023, the number of turbines in their suitable areas has remained high, covering 76.56% and 92.95% of the turbines built in 2011, and 83.20% and 94.89% of the turbines built in 2023.

Concerning the Griffon Vulture, at present, the south-east Sardinia area remains undisturbed by wind energy developments (Fig. S6). In contrast, the north-west area presented 25 wind turbines in 2011, while the most recent 2023 update underscores that the number of operational turbines within the vultures' core range in this area has increased to 90 (Fig. 3).

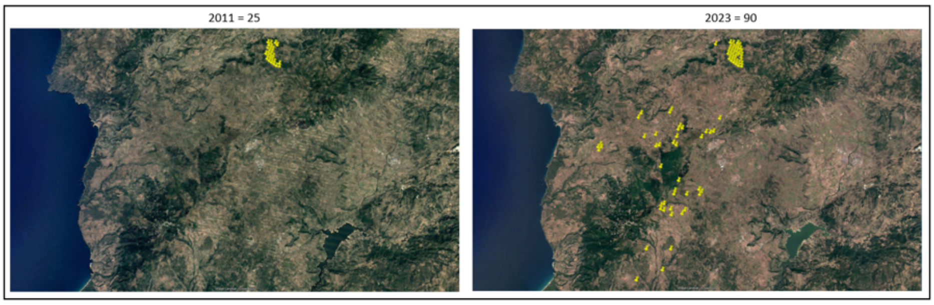

Figure 3: Side-by-side comparison of historical satellite imagery used to assess wind turbine development through time-lapse analysis. The left panel shows an older satellite image where few turbines were present (2011), while the right panel displays a more recent image where wind turbines have been installed at the same locations (2023).

The overlay with the cumulative habitat suitability raster for raptors (Fig. 5) revealed that a total of 100 km² within historically active areas fall within zones with suitability values equal to or greater than 0.50, accounting for 50.5% of the total area classified as historically active. In the case of the expansion zones, 335 km² fall within areas with suitability values equal to or greater than 0.50, representing 73.3% of the total area identified for expansion.

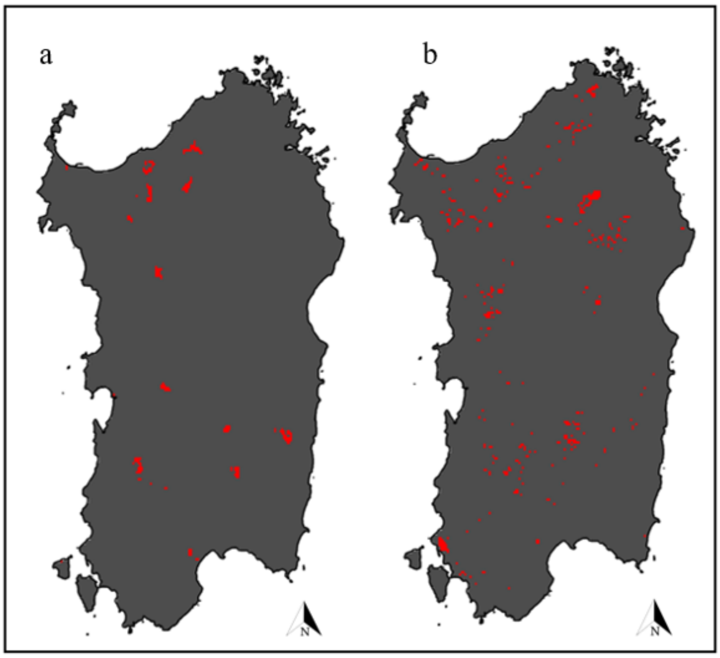

Figure 4: Spatiotemporal patterns of wind energy development on the island based on normalised wind turbine density rasters from 2011 and 2023. Historically active areas (panel a) indicate cells where wind energy infrastructure was already present in 2011 and remained active in 2023. Expansion zones (panel b) correspond to cells where wind energy activity emerged in 2023 but was absent in 2011, representing areas of recent wind farm expansion during the past decade.

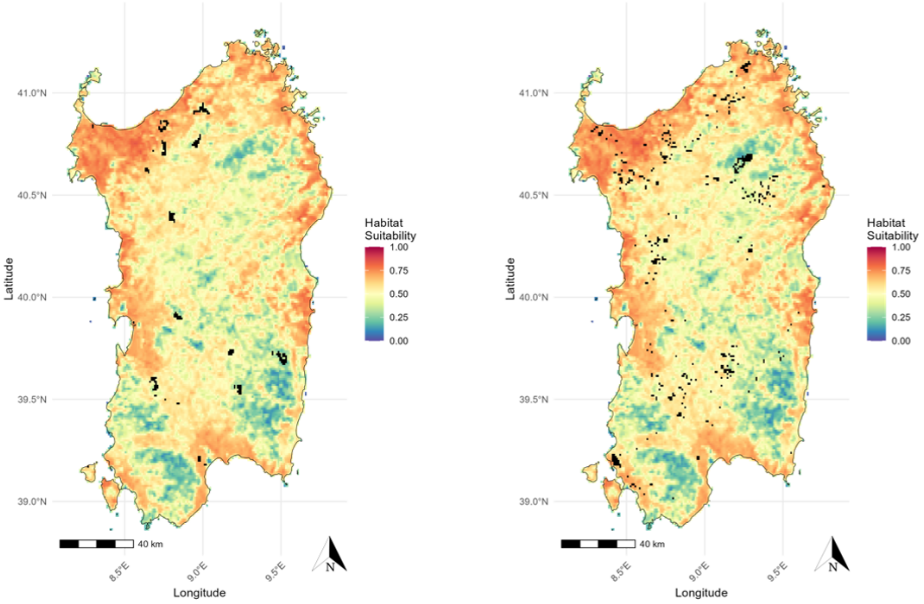

Figure 5: Cumulative habitat suitability map for seven raptor species in Sardinia, based on Species Distribution Models (SDMs), overlaid with historically active areas (left side), and expansion zones (right side).

4 Discussion

This study offers a novel perspective about the added value of combining the spatially and temporally variable assessment of turbine expansion with habitat suitability modelling. Our approach allowed us to: (i) identify spatial and temporal patterns in wind farm expansion, and (ii) quantify the overlap between installed wind turbines and areas of high cumulative suitability for multiple species of vulnerable raptors.

While the number of turbines in Sardinia increased by 2.18 times between 2011 and 2023, their construction did not occur uniformly across the island but was clustered in space and time (Fig. 1). While some areas have hosted wind turbines since at least 2011, others foresaw a more recent expansion of wind farms.

The distinction between historically active areas and expansion zones highlights two different phases in the development of wind energy in Sardinia, with important implications for i) environmental impact assessment, ii) mitigation measures aimed at reducing collisions between raptors and wind turbines, iii) spatial planning and iii) future research on how wind energy development can affect biological communities in the Mediterranean In terms of environmental impact assessment, the distinction between historically active areas and expansion zones calls for two different assessments aimed at filling two different knowledge gaps.

Historically active areas correspond to the island's earliest wind energy installations and have remained operational for over 13 years. This long-term presence implies that several sensitive species of birds and bats have been exposed to prolonged disturbance and/or mortality.

This could in turn have resulted into a long-term reduction of their fitness, due to behavioural changes, stress and/or mortality (Duriez et al., 2023; May et al., 2015), and in the creation of source-sink systems (Grainger-Hunt et al., 2017) or in the permanent alteration of entire assemblages of species(Fernández-Bellon et al., 2018).

Environmental impact assessment for wind farms in historically active areas should therefore focus on quantifying the long-term impact of wind turbines on individual fitness and population viability dynamics. Conversely, raptors in expansion zones, particularly those living in areas where wind farms developed after 2020, are still probably affected by increased disturbance and habitat modifications associated with the construction phase of wind farms (Schöll Nopp-Mayr, 2021).

Environmental impact assessment for wind farms in expansion zones should therefore prioritize the detection of short-term impacts on vulnerable species. In this context, historically active areas should represent the highest priority for monitoring, as they provide the ideal conditions to assess the prolonged impact of wind energy infrastructure on habitats and wildlife. Data gathered from these areas can also serve as a reference for predicting the future impact of wind turbines, for those areas where their construction was more recent.

Expansion zones, while more recent in origin, also warrant attention, particularly when located near historically active clusters. In such cases, the cumulative effect may be amplified by the spatial overlap of pressures. Expansion zones situated farther from existing developments may offer opportunities to assess wind energy impacts in previously undisturbed environmental contexts.

The overlap between these areas and the cumulative habitat suitability raster showed that a significant proportion of new wind farm installations falls within zones of high suitability for raptors. This finding is particularly relevant from a risk management perspective, as it suggests increasing exposure of sensitive species to potential interference from wind energy infrastructure.

Within this framework, the multi-species approach adopted represents a key element. Integrating the spatial suitability of multiple raptor species allows for the identification of areas that are potentially more sensitive, overcoming the limitations of single-species assessments. The resulting cumulative output provides a synthetic map of areas with the highest ecological vulnerability, which can serve as an operational tool to guide monitoring efforts and further investigation.

In particular, zones where wind energy expansion overlaps with high multi-species suitability should be considered priority targets for further assessments of collision risk and for the design of site-specific mitigation measures.

From an operational perspective, we recommend a graduated monitoring strategy, prioritising among the suitable areas: (i) historically active areas, (ii) expansion zones adjacent to these areas, and (iii) expansion zones located further away. This approach would optimise the use of available resources while enabling data collection across a spatio-temporal gradient of wind energy impact.

Species-specific analyses underscore the urgency of these findings. The Red Kite shows a marked preference for areas increasingly affected by wind energy development. This spatial overlap raises major concerns: due to their soaring flight and foraging behaviour, medium-sized raptors like the Red Kite are highly susceptible to collisions (Mattsson et al., 2022). Collision risk models suggest that even moderate mortality rates can threaten the viability of small, declining populations (Schaub, 2012) such as the Critically Endangered Red Kite in Sardinia. A forthcoming study (De Rosa et al., 2025) confirms that many breeding and wintering areas in Sardinia are under growing pressure from turbine expansion.

Similarly, the Golden Eagle is vulnerable to both collisions and habitat displacement. In places like California's Altamont Pass, turbine collisions kill dozens of individuals annually (Smallwood Thelander, 2008), and similar risks have been reported in Europe (Fielding et al., 2021). In Sardinia, increasing turbine density since 2011 has intensified threats. Even a few collisions can reduce foraging efficiency and reproductive success, compromising long-term survival (Hunt et al., 2017).

Generalist species such as the Common Buzzard and Common Kestrel, although listed as Least Concern, show broad overlap with wind turbines. The Common Buzzard, for instance, occupies areas with over 90% turbine coverage. Evidence from Italy and Germany points to both behavioural avoidance and high mortality rates (De Lisio et al., 2011; Grünkorn et al., 2017), suggesting potential long-term effects.

The Griffon Vulture represents a particularly critical case. This large, soaring scavenger depends on thermal currents and often flies in groups through high-risk areas near turbines (Arrondo et al., 2020).

Furthermore, the impact of wind energy development in Sardinia varies by area. The southern-east area is currently free from turbines. In contrast, the northwestern area has seen substantial expansion (Cerri et al., 2023), since the number of operational turbines rose from 25 in 2011 to 90 in 2023, significantly increasing the risk of turbine-vulture interactions (Fig. 3).

Moreover, Griffon Vultures exhibit poor avoidance behaviour: unlike species such as the Golden Eagle, which tend to avoid turbines and thus experience functional habitat loss (Fielding et al., 2021), Griffon Vulture frequently fly through high-risk areas without evasive action.

In some European regions, advanced strategies are already in place to mitigate bird collision risk with wind turbines, such as on-demand turbine shutdown in the presence of vulnerable species. In Andalusia, for example, a year-round surveillance system allows for the temporary shutdown of turbines when hazardous situations are detected, reducing Griffon Vulture mortality by around 50% with a negligible energy loss (De Lucas et al., 2012). Alongside human observers, automated systems such as radar and smart cameras are increasingly used to detect birds in real time and trigger immediate responses.

In Sardinia, where there is significant overlap between areas of high suitability for soaring raptors and zones targeted for wind energy expansion, active mitigation strategies of this kind are still lacking. Risk management largely relies on static, pre-construction assessments, with no dynamic monitoring during turbine operation. Currently, neither real-time surveillance nor automated detection technologies are in place.

Overall, this study underscores the necessity of integrating spatial and temporal data on wind energy development with multi-species habitat models to inform conservation strategies. The clear spatial clustering of turbines in areas of high cumulative suitability, coupled with uneven temporal dynamics, highlights the need for more proactive and adaptive management approaches.

Periodic updates to turbine databases, regular monitoring of raptor populations, and the adoption of avoidance-based siting criteria are essential steps to mitigate long-term impacts. As renewable energy continues to expand across the Mediterranean, Sardinia emerges as a critical case study, illustrating both the risks of poorly planned development and the value of biodiversity-informed decision-making.

This study presents an unprecedented spatiotemporal assessment of wind energy development over a 13-year period, focusing on eight raptor species of conservation concern. Unlike previous research, which has often examined shorter timeframes or single species, our approach offers a more comprehensive understanding of spatial and temporal trends, revealing how wind energy expansion can proceed in a non-uniform manner and potentially conflict with biodiversity conservation goals.

The combination of a long-term perspective and a multi-species framework represents a significant methodological advancement, providing more robust insights for spatial planning and for evaluating the cumulative risks associated with wind energy development. Our findings reinforce the urgent need for regularly updated data, rigorous environmental assessments, and the integration of ecological monitoring into decision-making processes, to support genuine coexistence between energy transition and nature conservation.

In this context, our work provides a solid scientific foundation for the development of more effective mitigation strategies, that address not only the siting of new wind farms, but also the adaptive management of risks for the most vulnerable species.

References

[1] Aiello-Lammens, M. A., Boria, R. A., Radosavljevic, A., Vilela, B., & Anderson, R. P. (2015). spThin: an R package for spatial thinning of species occurrence records for use in ecological niche models. *Ecography*, 38, 000–000. https://doi.org/10.1111/ecog.01132

[2] Arnett, E. B., & May, R. F. (2016). Mitigating wind energy impacts on wildlife: approaches for multiple taxa. *Human-Wildlife Interactions*, 10(1), 28–41. https://www.jstor.org/stable/24874154

[3] Arrondo, E., Sanz-Aguilar, A., Perez-García, J. M., Cortés-Avizanda, A., Sánchez-Zapata, J. A., & Donázar, J. A. (2020). Landscape anthropization shapes the survival of a top avian scavenger. *Biodiversity and Conservation*, 29, 1411–1425. https://doi.org/10.1007/s10531-020-01942-6

[4] Bailey, S. A., Haines-Young, R. H., & Watkins, C. (2002). Species presence in fragmented landscapes: modelling of species requirements at the national level. *Biological Conservation*, 108(3), 307–316. https://doi.org/10.1016/S0006-3207(02)00119-2

[5] Barbet-Massin, M., Jiguet, F., Albert, C. H., & Thuiller, W. (2012). Selecting pseudo-absences for species distribution models: How, where and how many? *Methods in Ecology and Evolution*, 3(2), 327–338. https://doi.org/10.1111/j.2041-210X.2011.00172.x

[6] Barrios, L., & Rodriguez, A. (2004). Behavioural and environmental correlates of soaring-bird mortality at on-shore wind turbines. *Journal of Applied Ecology*, 41(1), 72–81. https://doi.org/10.1111/j.1365-2664.2004.00876.x

[7] Barve, N., Barve, V., Jiménez-Valverde, A., Lira-Noriega, A., Maher, S. P., Peterson, A. T., & Villalobos, F. (2011). The crucial role of the accessible area in ecological niche modeling and species distribution modeling. *Ecological Modelling*, 222(11), 1810–1819. https://doi.org/10.1016/j.ecolmodel.2011.02.011

[8] Bildstein, K. L. (2022). *Vultures of the world: essential ecology and conservation*. Cornell University Press. https://www.cornellpress.cornell.edu/book/9781501761614/vultures-of-the-world/

[9] Bose, A., Dürr, T., Klenke, R. A., & Henle, K. (2020). Correction: Predicting strike susceptibility and collision patterns of the common buzzard at wind turbine structures in the federal state of Brandenburg, Germany. *PLOS ONE*, 15(8), e0238269. https://doi.org/10.1371/journal.pone.0238269

[10] Braunisch, V., Coppes, J., Bächle, S., & Suchant, R. (2015). Underpinning the precautionary principle with evidence: A spatial concept for guiding wind power development in endangered species' habitats. *Journal for Nature Conservation*, 24, 31–40. https://doi.org/10.1016/j.jnc.2015.01.003

[11] Bright, J., Langston, R., Bullman, R., Evans, R., Gardner, S., & Pearce-Higgins, J. (2008). Map of bird sensitivities to wind farms in Scotland: a tool to aid planning and conservation. *Biological Conservation*, 141(9), 2342–2356. https://doi.org/10.1016/j.biocon.2008.06.029

[12] Carrete, M., Sanchez-Zapata, J. A., Benítez, J. R., Lobón, M., Montoya, F., & Donázar, J. A. (2012). Mortality at wind-farms is positively related to large-scale distribution and aggregation in griffon vultures. *Biological Conservation*, 145, 102–108. https://doi.org/10.1016/j.biocon.2011.10.017

[13] Cerri, J., Costantino, C., De Rosa, D., Banic, D. A., Urgeghe, G., Fozzi, I., Echeverria, J., Aresu, M., & Berlinguer, F. (2024a). Widely used datasets of wind energy infrastructures can seriously underestimate onshore turbines in the Mediterranean. *Biological Conservation*, 300, 110870. https://doi.org/10.1016/j.biocon.2024.110870

[14] Cerri, J., Fozzi, I., Costantino, C., Banič, D. A., De Rosa, D., Echeverria, J., & Berlinguer, F. (2024b). Different sources of wind turbine data produce sharp differences in collision risk estimates for foraging vultures. https://doi.org/10.32942/X2JS8J

[15] Cerri, J., Fozzi, I., De Rosa, D., Aresu, M., Apollonio, M., & Berlinguer, F. (2023). Griffon Vulture movements are concentrated around roost and supplementary feeding stations: implications for wind energy development on Mediterranean islands. *Global Ecology and Conservation*, 47, e02651. https://doi.org/10.1016/j.gecco.2023.e02651

[16] De Lisio, L., Di Perna, D., & Loy, A. (2011). Wind Farm and raptors: a case study in Molise (southern Italy): red kite (Milvus milvus) and common buzzard (Buteo buteo) in SIC "La Montagnola Molisana" IT7212135. *Abstracts of the II International Congress Problematic Wildlife: Conservation and Management*, Genazzano, Roma, 3–5 febbraio 2011.

[17] de Lucas, M., Ferrer, M., Bechard, M. J., & Muñoz, A. R. (2012). Griffon vulture mortality at wind farms in southern Spain: Distribution of fatalities and active mitigation measures. *Biological Conservation*, 147(1), 184–189. https://doi.org/10.1016/j.biocon.2011.12.029

[18] de Lucas, M., Janss, G. F., Whitfield, D. P., & Ferrer, M. (2008). Collision fatality of raptors in wind farms does not depend on raptor abundance. *Journal of Applied Ecology*, 45(6), 1695–1703. https://doi.org/10.1111/j.1365-2664.2008.01549.x

[19] De Rosa, D., Fozzi, I., Costantino, C., Uleri, G. I., Caddeo, G., Aresu, M., Cerri, J., & Berlinguer, F. (2025). Which future for Red Kite (Milvus milvus) in Sardinia? An analysis of the future scenario of developing wind farms. *Bird Study*, (accepted).

[20] Drewitt, A. L., & Langston, R. H. (2006). Assessing the impacts of wind farms on birds. *Ibis*, 148, 29–42. https://doi.org/10.1111/j.1474-919X.2006.00516.x

[21] Duriez, O., Pilard, P., Saulnier, N., Boudarel, P., & Besnard, A. (2023). Wind farm collisions in medium-sized raptors: even increasing populations can suffer strong demographic impacts. *Animal Conservation*, 26(2), 264–275. https://doi.org/10.1111/acv.12818

[22] Elith, J., Graham, C. H., Anderson, R. P., Dudík, M., Ferrier, S., Guisan, A., Hijmans, R. J., Huettmann, F., Leathwick, J. R., Lehmann, A., Li, J., Lohmann, L. G., Loiselle, B. A., Manion, G., Moritz, C., Nakamura, M., Nakazawa, Y., Overton, J. M., Peterson, A. T., Phillips, S. J., Richardson, K., Scachetti-Pereira, R., Schapire, R. E., Soberón, J., Williams, S., Wisz, M. S., & Zimmermann, N. E. (2006). Novel methods improve prediction of species' distributions from occurrence data. *Ecography*, 29, 129–151. https://doi.org/10.1111/j.2006.0906-7590.04596.x

[23] Fernández-Bellon, D., Wilson, M. W., Irwin, S., & O'Halloran, J. (2019). Effects of development of wind energy and associated changes in land use on bird densities in upland areas. *Conservation Biology*, 33(2), 413–422. https://doi.org/10.1111/cobi.13239

[24] Ferrer, M., Alloing, A., Baumbush, R., & Morandini, V. (2022). Significant decline of Griffon Vulture collision mortality in wind farms during 13 years of a selective turbine stopping protocol. *Global Ecology and Conservation*, 38, e02203. https://doi.org/10.1016/j.gecco.2022.e02203

[25] Fielding, A. H., Anderson, D., Benn, S., Dennis, R., Geary, M., Weston, E., & Whitfield, D. P. (2021). Non-territorial GPS-tagged golden eagles Aquila chrysaetos at two Scottish wind farms: Avoidance influenced by preferred habitat distribution, wind speed and blade motion status. *PLOS ONE*, 16(8), e0254159. https://doi.org/10.1371/journal.pone.0254159

[26] Frick, W. F., Baerwald, E. F., Pollock, J. F., Barclay, R. M. R., Szymanski, J. A., Weller, T. J., McGuire, L. P., et al. (2017). Fatalities at wind turbines may threaten population viability of a migratory bat. *Biological Conservation*, 209, 172–177. https://doi.org/10.1016/j.biocon.2017.02.023

[27] Garvin, J. C., Jennelle, C. S., Drake, D., & Grodsky, S. M. (2011). Response of raptors to a windfarm. *Journal of Applied Ecology*, 48(1), 199–209. https://doi.org/10.1111/j.1365-2664.2010.01912.x

[28] Global Wind Energy Council. (2020). *Global Wind Report 2019*. https://gwec.net/global-wind-report-2019/

[29] Grainger Hunt, W., Wiens, J. D., Law, P. R., Fuller, M. R., Hunt, T. L., Driscoll, D. E., & Jackman, R. E. (2017). Quantifying the demographic cost of human-related mortality to a raptor population. *PLOS ONE*, 12(2), e0172232. https://doi.org/10.1371/journal.pone.0172232

[30] Grünkorn, T., Blew, J., Krüger, O., Potiek, A., Reichenbach, M., von Rönn, J., et al. (2017). A large-scale, multispecies assessment of avian mortality rates at land-based wind turbines in northern Germany. In Köppel, J. (Ed.), *Wind Energy and Wildlife Interactions: Presentations from the CWW2015 Conference* (pp. 43–64). Springer International Publishing. https://doi.org/10.1007/978-3-319-51272-3_3

[31] Guisan, A., Thuiller, W., & Zimmermann, N. E. (2017). *Habitat suitability and distribution models: with applications in R*. Cambridge University Press.

[32] Hirzel, M., Diwan, A., & Hind, M. (2004). Pointer analysis in the presence of dynamic class loading. In *European Conference on Object-Oriented Programming* (pp. 96–122). Berlin, Heidelberg: Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-540-24851-4_5

[33] Hunt, W. G. (2002). Golden eagles in a perilous landscape: Predicting the effects of mitigation for energy-related mortality. Report to the California Energy Commission, PIER Grant No. 500-97-4033 to the University of California, Santa Cruz, CA. https://tethys.pnnl.gov/sites/default/files/publications/Hunt-2002.pdf

[34] Hunt, W. G., Wiens, J. D., Law, P. R., Fuller, M. R., Hunt, T. L., Driscoll, D. E., & Jackman, R. E. (2017). Quantifying the demographic cost of human-related mortality to a raptor population. *PLOS ONE*, 12(2), e0172232. https://doi.org/10.1371/journal.pone.0172232

[35] Huso, M., Dalthorp, D., Miller, T. J., & Bruns, D. (2016). Wind energy development: methods to assess bird and bat fatality rates post-construction. *Human-Wildlife Interactions*, 10(1), 62–70. https://www.jstor.org/stable/24874157

[36] Janss, G. F., & Ferrer, M. (2000). Rate of bird collision with power lines: effects of conductor-marking and static wire-marking (Tasa de Choques por Parte de Aves con Líneas del Tendido Eléctrico: Efecto de Marcadores de Conducción y Marcadores de Estática). *Journal of Field Ornithology*, 71(1), 8–17. https://www.jstor.org/stable/4514280

[37] Janss, G. F., Lucas, M. D., Whitfield, D. P., Lazo, A., & Ferrer, M. (2010). The precautionary principle and wind-farm planning in Andalucía. Retrieved from `https://digital.csic.es/bitstream/10261/59163/1/biolcon.pdf`

[38] Jonasson, K. A., Adams, A. M., Brokaw, A. F., Whitby, M. D., O'Mara, M. T., & Frick, W. F. (2024). A multisensory approach to understanding bat responses to wind energy developments. *Mammal Review*. https://doi.org/10.1111/mam.12340

[39] Katzner, T. E., Brandes, D., Miller, T., Lanzone, M., Maisonneuve, C., Tremblay, J. A., et al. (2012). Topography drives migratory flight altitude of golden eagles: implications for on-shore wind energy development. *Journal of Applied Ecology*, 49(5), 1178–1186. https://doi.org/10.1111/j.1365-2664.2012.02185.x

[40] Kranstauber, B., Kays, R., LaPoint, S. D., Wikelski, M., & Safi, K. (2012). A dynamic Brownian bridge movement model to estimate utilization distributions for heterogeneous animal movement. *Journal of Animal Ecology*, 81(4), 738–746. https://doi.org/10.1111/j.1365-2656.2012.01955.x

[41] Kriebel, D., & Tickner, J. (2001). Reenergizing public health through precaution. *American Journal of Public Health*, 91(9), 1351–1355. https://doi.org/10.2105/AJPH.91.9.1351

[42] Krüeger, S. C., Allan, D. G., Jenkins, A. R., & Amar, A. (2014). Trends in territory occupancy, distribution and density of the Bearded Vulture *Gypaetus barbatus meridionalis* in southern Africa. *Bird Conservation International*, 24(2), 162–177. https://doi.org/10.1017/S0959270913000440

[43] Larsen, J. K., & Clausen, P. (2002). Potential wind park impacts on whooper swans in winter: the risk of collision. *Waterbirds*, 25(3), 327–330. https://www.jstor.org/stable/1522370

[44] Madders, M., & Whitfield, D. P. (2006). Upland raptors and the assessment of wind farm impacts. *Ibis*, 148, 43–56. https://doi.org/10.1111/j.1474-919X.2006.00506.x

[45] Manel, S., Dias, J.-M., & Ormerod, S. J. (1999). Comparing discriminant analysis, neural networks and logistic regression for predicting species distributions: a case study with a Himalayan river bird. *Ecological Modelling*, 120, 337–347. https://doi.org/10.1016/S0304-3800(99)00113-1

[46] Marques, A. T., Batalha, H., Rodrigues, S., Costa, H., Pereira, M. J. R., Fonseca, C., et al. (2014). Understanding bird collisions at wind farms: An updated review on the causes and possible mitigation strategies. *Biological Conservation*, 179, 40–52. https://doi.org/10.1016/j.biocon.2014.08.017

[47] Martin, G. R., Portugal, S. J., & Murn, C. P. (2012). Visual fields, foraging and collision vulnerability in *Gyps* vultures. *Ibis*, 154(3), 626–631. https://doi.org/10.1111/j.1474-919X.2012.01227.x

[48] Mattsson, B. J., Mateo-Tomás, P., Aebischer, A., Rösner, S., Kunz, F., Schöll, E. M., et al. (2022). Enhancing monitoring and transboundary collaboration for conserving migratory species under global change: The priority case of the red kite. *Journal of Environmental Management*, 317, 115345. https://doi.org/10.1016/j.jenvman.2022.115345

[49] May, R. F. (2015). A unifying framework for the underlying mechanisms of avian avoidance of wind turbines. *Biological Conservation*, 190, 179–187. https://doi.org/10.1016/j.biocon.2015.06.004

[50] Pearce-Higgins, J. W., Stephen, L., Douse, A., & Langston, R. H. (2012). Greater impacts of wind farms on bird populations during construction than subsequent operation: results of a multi-site and multi-species analysis. *Journal of Applied Ecology*, 49(2), 386–394. https://doi.org/10.1111/j.1365-2664.2012.02110.x

[51] Pennycuick, C. J. (1998). Field observations of thermals and thermal streets, and the theory of cross-country soaring flight. *Journal of Avian Biology*, 29(1), 33–43. https://doi.org/10.2307/3677338

[52] Pennycuick, C. J., Farner, D. S., King, J. R., & Parkes, K. C. (1975). Mechanics of flight. In D. S. Farner, J. R. King, & K. C. Parkes (Eds.), *Avian Biology* (Vol. 5, pp. 1–75). Academic Press.

[53] Poessel, S. A., Brandt, J., Mendenhall, L., Braham, M. A., Lanzone, M. J., McGann, A. J., & Katzner, T. E. (2018). Flight response to spatial and temporal correlates informs risk from wind turbines to the California Condor. *The Condor: Ornithological Applications*, 120(2), 330–342. https://doi.org/10.1650/CONDOR-17-100.1

[54] Prasad, M., Zisserman, A., Fitzgibbon, A., Kumar, M. P., & Torr, P. H. (2006). Learning class-specific edges for object detection and segmentation. In *Computer Vision, Graphics and Image Processing: 5th Indian Conference, ICVGIP 2006, Madurai, India, December 13–16, 2006. Proceedings* (pp. 94–105). Springer Berlin Heidelberg. https://doi.org/10.1007/11949619_9

[55] Raganella Pellicciomi, E., Di Vittorio, M., Avramo, V., Nardelli, R., Spina, F., & Serra, L. (2024). Conflitti con le attività umane o minacce per la conservazione? Il caso dell'Aquila di Bonelli in Sardegna. Manoscritto inedito.

[56] Rushworth, I., & Krüger, S. (2014). Wind farms threaten southern Africa's cliff-nesting vultures. *Ostrich*, 85(1), 13–23. https://doi.org/10.2989/00306525.2014.913211

[57] Rydell, J., Bach, L., Dubourg-Savage, M. J., Green, M., Rodrigues, L., & Hedenström, A. (2010). Mortality of bats at wind turbines links to nocturnal insect migration? *European Journal of Wildlife Research*, 56, 823–827. https://doi.org/10.1007/s10344-010-0444-3

[58] Schaub, M. (2012). Spatial distribution of wind turbines is crucial for the survival of red kite populations. *Biological Conservation*, 155, 111–118. https://doi.org/10.1016/j.biocon.2012.06.021

[59] Schöll, E. M., & Nopp-Mayr, U. (2021). Impact of wind power plants on mammalian and avian wildlife species in shrub-and woodlands. *Biological Conservation*, 256, 109037. https://doi.org/10.1016/j.biocon.2021.109037

[60] Segurado, P., & Araújo, M. B. (2004). An evaluation of methods for modelling species distributions. *Journal of Biogeography*, 31, 1555–1568. https://doi.org/10.1111/j.1365-2699.2004.01076.x

[61] Seo, C., Thorne, J. H., Hannah, L., & Thuiller, W. (2009). Scale effects in species distribution models: implications for conservation planning under climate change. *Biology Letters*, 5(1), 39–43. https://doi.org/10.1098/rsbl.2008.0476

[62] Sergio, F., & Hiraldo, F. (2008). Intraguild predation in raptor assemblages: a review. *Ibis*, 150, 132–145. https://doi.org/10.1111/j.1474-919X.2008.00786.x

[63] Shenk, T. (2009). Tabelle sullo stato di conservazione degli uccelli in Sardegna. In M. Aresu, A. Fozzi, & B. Massa (Eds.), *Una vita per la natura: Omaggio a Helmar Schenk* (pp. 203–215). Unione Sarda.

[64] Shrestha, N. (2020). Detecting multicollinearity in regression analysis. *American Journal of Applied Mathematics and Statistics*, 8(2), 39–42. https://doi.org/10.12691/ajams-8-2-1

[65] Smallwood, K. S., & Thelander, C. (2008). Bird mortality in the Altamont Pass wind resource area, California. *The Journal of Wildlife Management*, 72(1), 215–223. https://doi.org/10.2193/2007-032

[66] Stockwell, D. R., & Peterson, A. T. (2002). Effects of sample size on accuracy of species distribution models. *Ecological Modelling*, 148(1), 1–13. https://doi.org/10.1016/S0304-3800(01)00388-X

[67] Venter, J. A., Martens, F. R., & Wolter, K. (2019). Conservation buffer sizes derived from movement data of adult Cape vultures (*Gyps coprotheres*) in South Africa. *African Zoology*, 54(2), 115–118. https://doi.org/10.1080/15627020.2019.1600428

[68] Willsteed, E. A., Jude, S., Gill, A. B., & Birchenough, S. N. (2018). Obligations and aspirations: a critical evaluation of offshore wind farm cumulative impact assessments. *Renewable and Sustainable Energy Reviews*, 82, 2332–2345. https://doi.org/10.1016/j.rser.2017.08.079

[69] Woolf, A., Nielsen, C. K., Fredrickson, L., & Woolf, M. (2002). Statewide modeling of bobcat (*Lynx rufus*) habitat in Illinois, USA. *Biological Conservation*, 104, 191–198. https://doi.org/10.1016/S0006-3207(01)00164-1

[70] Zimmerling, J., Pomeroy, A., d'Entremont, M., & Francis, C. (2013). Canadian estimate of bird mortality due to collisions and direct habitat loss associated with wind turbine developments. *Avian Conservation and Ecology*, 8(2). https://doi.org/10.5751/ACE-00609-080210