



Fine-scale collision risk mapping and validation with long-term mortality data reveal current and future wind energy development impact on sensitive species

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ABSTRACT

The ongoing demand for renewable energy has boosted the development of wind farms worldwide. Given the impact these facilities have on flying species, a spatially explicit assessment of collision risk in vulnerable species is needed to guide management actions and prioritise areas for installing these infrastructures. We used GPS-tracking data of 127 adult and 50 juvenile griffon vultures in peninsular Spain gathered between 2014 and 2022 to evaluate factors influencing vulnerability and exposure and predict collision risk. We validate the observed collision risk with recorded long-term mortality data (1999–2022) at regional and wind farm scales and evaluate the estimated impact of current and future turbine facilities. Our results showed that overall food availability increases vulnerability and exposure, whilst distance to nesting areas and the presence of conspecifics decreased both vulnerability and exposure in adults and juveniles, respectively. Our maps revealed that 19% and 10% of the Spanish peninsular area had a high collision risk for adults and juveniles, respectively. Importantly, the number of turbine casualties was positively related to collision risk at the regional and wind farm scale and ~ 18 of the breeding population lies within high collision risk areas. Moreover, the areas with the highest risk of collision also have the highest number of turbines and largely overlap with areas suitable for developing new wind farms. Our study highlights the need to reduce collision risk mapping uncertainties by validating model outputs with actual mortality data. Moreover, it emphasises the urgent need for spatial planning of wind energy development, searching for safer alternatives for biodiversity. This approach undoubtedly serves as a tool to define “not go to” areas for installing new turbines for one of the most sensitive species.

1. Introduction

Wind energy production has experienced a continuous expansion worldwide, with an increase of almost 30% in the last decade (IRENA, 2020). While this energy has been able to meet the demand of five million people worldwide, it has proven to have undesirable effects.

Numerous studies have highlighted the diverse impacts of wind farms on biodiversity, ranging from noise pollution (Teff-Seker et al., 2022; Gómez-Catasús et al., 2022), habitat alterations (Agha et al., 2020; May et al., 2021; Ellerbrok et al., 2022), or changes in community composition (Falavigna et al., 2020) to more extreme ones such as increased non-natural mortality of bats and birds due to collision with turbines

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(Carrete et al., 2009; Jager et al., 2021).

Increases in non-natural mortality from turbine collision in long-lived species such as soaring raptors have attracted the attention of governments and conservationists (Katzner et al., 2019; Serrano et al., 2020) because of their direct effect on the long-term viability of their populations (Carrete et al., 2009; Sanz-Aguilar et al., 2015; Conkling et al., 2022; Duriez et al., 2022). In recent years, several studies have addressed the collision risk of these species in wind farms using radar technologies (Aschwanden et al., 2018; Cohen et al., 2022) and, more recently, movement data (Péron et al., 2017; Cervantes et al., 2023). New tracking technologies have enabled ecologists to develop spatially explicit collision risk maps using GPS-tracked birds at continental scales (see Péron et al., 2017; Thaxter et al., 2019; Murgatroyd et al., 2021; Gauld et al., 2022; Cervantes et al., 2023). Most of these studies use species flight altitude to delineate the collision risk of moving individuals in relation to the vertical sweep of turbine blades, considering avoidance behaviours, speed of movement and intensity of use of different areas (Masden and Cook, 2016). Other studies have also shown how some atmospheric factors (e.g., uplift presence and wind, Péron et al., 2017), habitat (Tikkanen et al., 2018) or species characteristics (i.e., migratory behaviour) can affect collision risk (Hüppop et al., 2006; Thaxter et al., 2017). Despite their interest, these studies generally do not separately assess vulnerability (measured as the bird flight altitude) and exposure (i.e., the frequency with which birds use the areas where these facilities are located, obtained from bird movements) (but see Thaxter et al., 2017). This distinction is important as both aspects - which ultimately determine the impact of wind farms - can be modulated by different combinations of factors, such as uplift presence, habitat characteristics, food availability, or intrinsic characteristics of species (e.g., flight altitude during foraging, commuting or migration; Schwemmer et al., 2023) or individuals (e.g., age or social attraction) (see Cervantes et al., 2023).

The griffon vulture (*Gyps fulvus*) is a gregarious and colonial species still relatively abundant and very sensitive to collision in wind farms due to the use of air currents and its sensory characteristics (Martin et al., 2012; Potier, 2020). The species has shown one of the highest mortality rates recorded in wind farms, with the death of ca. 9000 vultures (850 per year) recorded during the last two decades only in Spain (Pérez-García and Serrano, 2023). To these mortality events must be added other ongoing threat factors such as collision with other infrastructures, electrocution, lead poisoning or pharmaceutical drugs (Green et al., 2016; Blanco et al., 2016; Arrondo et al., 2020). The griffon vulture is one of the species for which the largest ranges of movements have been described (up to 10,000 km², with individuals travelling ca. 900 km; Morant et al., 2023a, 2023b; Spiegel et al., 2013). Thus, the spatial planning of renewable energies is challenging, especially in a scenario of growth of these infrastructures (Serrano et al., 2020). It is, therefore, urgent to develop tools that consider the factors that influence the vulnerability and exposure of this species to infer mortality hotspots that lack monitoring and assess the suitability of areas where new wind farms are planned. Studying areas at risk of collision using griffon vultures as a model will not only help determine safe zones for this vulnerable species but, due to its good performance as a biomonitor of collisions, could be applied to other concerned species with limited data or with difficulties in monitoring (Sebastián-González et al., 2018).

In this study, we evaluated for the first time the effect of environmental constraints (e.g., uplift presence, habitat type, orography, and food availability), conspecific attraction (e.g., distance and size of the nearest colony and distance to the breeding nest) on the vulnerability and exposure of juvenile and adult griffon vultures to wind farms. For this purpose, we used data from 177 adult and juvenile GPS-tagged vultures monitored between 2014 and 2022 in peninsular Spain, home to 90% of European griffon vulture populations (Margalida et al., 2010). We expected flight suitability and food availability to influence the vulnerability and exposure of griffon vultures to wind farm turbines, as they take advantage of uplifts to prospect large areas at low altitudes to

find carrion. We also predicted that conspecific attraction would increase exposure near breeding colonies, as vultures use these areas more frequently (Carrete et al., 2012; Harel et al., 2017). We then created spatially explicit predictions for both vulnerability and exposure of adults and juveniles to generate a single map that accounts for the multiplicative effects of these two parameters. Next, we validated the collision risk map with field data on griffon vulture fatalities at regional and wind farm scales. Finally, we estimated the current and future impact of wind farms using spatial data on current turbine density and wind power generation potential, in particular the percentage of the breeding population located in areas with different levels of collision risk.

2. Methods

2.1. Movement data

Between 2014 and 2022, we tagged 177 griffon vultures (127 adults and 50 juveniles) in five Iberian populations distributed across northern (Alto Ebro and Pyrenees: 63 and 26 individuals, respectively), central (Avila, Segovia and Castellon: 5, 20 and 1 individuals) and southern Spain (Cádiz and Cazorla: 20 and 42 individuals) (see Table S1 for details). Adults were captured with remotely activated cannon nets and cage traps, while juveniles were tagged in nests when they were non-fledgling chicks. Following Morant et al. (2023a, 2023b), populations were delimited considering the spatial distribution of breeding colonies and individual movements. Individuals were marked with plastic and metal alphanumeric rings and equipped with solar-powered GPS/GSM transmitters (Ecotone <https://ecotone-telemetry.com/en>, Ornitela <https://www.ornitela.com/> and e-Obs <https://e-obs.de/>). The total weight of transmitters and rings did not exceed 64 g, which represents <3% of the body weight of the individuals (Bodey et al., 2018). The age of individuals was estimated from plumage traits (Donazar, 1993; Zuberogoitia et al., 2013), while sex was determined by molecular sexing techniques (Fridolfsson and Ellegren, 1999).

Tracking devices included a GPS that provided geographic coordinates and were programmed to record fixes (i.e., GPS positions) at 5-10 min intervals from 1 h before sunrise to 1 h after sunset. The GPS data were incorporated into the Movebank online data repository (www.movebank.org). Data were standardised by resampling GPS fixed to 15 min whenever individuals had 10 min fix records scheduled. We, therefore, retained 5,659,502 fixes (see Table S1 for details).

We selected GPS fixes recorded in Peninsular Spain for all tagged birds. We then classified each location as corresponding to a flying or non-flying bird using the altitude and ground speed recorded by the devices. First, we obtained for each fix its altitude above ground considering the altitude provided by the devices and the height of the earth's surface at each point (cell size of 20 × 20 m; IGNE, 2021), removing unrealistic flight altitudes (e.g., values below 0 or above 10,000 m; see Arrondo et al., 2021 for a similar approach). Second, we considered 2.5 m/s as our above-ground speed threshold for flight locations (Schlaich et al., 2016; Arrondo et al., 2021; Gauld et al., 2022) to retain fixes with altitudes >5 m above ground that had a speed equal or greater than this value ($n = 1,547,886$).

2.2. Vulnerability and exposure estimation

We used movement data to estimate vulnerability and exposure on a grid of 20,356 5 × 5 km UTM cells representing a total area of 505,990 km², corresponding to the whole of peninsular Spain. We estimated vulnerability by classifying cell grids as at risk when they had at least one GPS location of a vulture flying at a height between 16 and 210 m (see also Gauld et al., 2022 and Cervantes et al., 2023 for a similar approach). This range corresponds to the height of wind turbines (i.e., hub heights plus blade lengths) in Spain (see the database at www.thewindpower.net/). Exposure was calculated by using bird tracks to

measure the frequency of use of each cell. Bird track was defined as the spatial trajectory of a given individual from the start of its monitoring until the last day on which GPS locations were recorded (in our case, May 2022). Cell use was estimated as the total number of times that an individual crossed a given cell, regardless of flight height.

2.3. Explanatory variables

We selected variables describing factors relevant to understanding the vulnerability and exposure of griffon vultures at wind farms. Conspicuous attraction was estimated as 1) the distance from the centroid of each grid cell to the nearest griffon vulture breeding colony and 2) the number of breeding pairs in the nearest griffon vulture breeding colony, obtained from [Del Moral and Molina \(2018\)](#). For adult birds, we also measured the distance from the centroid of each grid cell to the most frequently used nest. Food availability was estimated using the information of the five primary carrion resources (i.e., livestock, roadkill, big game hunting, predation and carrion from natural mortality) mapped at 10x10km ([Morant et al., 2023a,b](#)) and food from other frequently exploited anthropogenic sources such as dumpsites ([Arrondo et al., 2023](#)). Carrion abundances were converted into probabilities as a better proxy of carrion availability by using random forest models and depicting the generated probabilities (see Supp Mat Appendix A1 for probabilities calculations) ([Morant et al., 2023a,b](#)). To estimate the influence of dumpsites, we measured the distance from the centroid of each grid cell to the nearest dumpsite, obtained from [MITECO \(2018\)](#). We also obtained the probability of thermal uplift in a given cell from a specific model developed for the griffon vulture by [Scacco et al. \(2023\)](#), where suitability for soaring in each pixel is represented in a range between 0 and 1. We estimated the proportion of open habitats commonly used by vultures during foraging (i.e., shrublands, woody savannahs, savannahs, grasslands, croplands, mosaics of croplands and natural vegetation, barrens, and permanent wetlands) in a 10 km buffer around the centroid of each grid cell (see [Cervantes et al., 2023](#) for a similar approach). Land use types were obtained from 2014 to 2021 at a 500 m resolution from the MODISsp package ([Busetto and Ranghetti, 2016](#)). Finally, slope, terrain roughness, and aspect were obtained using a 20 m resolution digital elevation model of Spain ([IGNE, 2021](#)) and the “terrain” function implemented in the “raster” package ([Hijmans, 2020](#)).

2.4. Modelling procedures

We run two independent sets of models for vulnerability and exposure at the individual level using spatial predictive-process GLMMs (Generalised Linear Mixed Effects Models) using Template Model Builder (TMB) (hereafter sdmTMB; [Anderson et al., 2022](#)). In these models, estimation is performed via maximum marginal likelihood and random effects are integrated using the Laplace approximation. According to model requirements, we created a grid matrix (defining the minimum allowed distance between points in the units of X and Y, in km) by setting a cut-off of 5 km that matches the size of our grid cell (see [Anderson et al., 2022](#) for details). The grid matrix is an artificial set of neighbours over the study area that allows us to calculate the spatial autocorrelation between observations. We selected these models since they offer robust estimates while accounting for spatiotemporal autocorrelation, estimation is often faster than alternatives, accommodates a variety of distribution families, and can be easily constructed with a formula interface ([Anderson et al., 2022](#)).

Vulnerability was modelled using a binomial error distribution (0 = no risk, 1 = risk on each cell) and the logit link function, while exposure (number of times that an individual crossed a cell) was modelled using a Poisson distribution and a log link function. Independent variables were included as covariates, and vulture identity as a random term to account for pseudoreplication. To reduce collinearity, independent variables within each group that showed a Pearson correlation coefficient > 0.6 were removed ([Dormann et al., 2013](#)), retaining the one with the best

biological meaning in each case. This meant excluding aspect, terrain roughness and carrion from natural mortality as covariates in the models. All variables were scaled (i.e., mean centred) before being included in models (full model approach; [Whittingham et al., 2006](#)) to obtain standardised estimates of their effects. The full models for vulnerability and exposure, respectively, are:

$$\log\left(\frac{p}{1-p}\right) = \beta_0 + \beta_1 * \text{distance_to_nest} + \beta_2 * \text{colony_size} \\ + \beta_3 * \text{colony_distance} + \beta_4 * \text{Uplift} + \beta_5 * \text{distance_to_dumpsite} \\ + \beta_6 * \text{livestock_prob} + \beta_7 * \text{big_game_hunting_prob} \\ + \beta_8 * \text{roadkill_prob} + \beta_9 * \text{predation_prob} + \beta_{10} * \text{slope} \\ + \beta_{11} * \text{open_areas} + \alpha$$

$$\log(\lambda) = \beta_0 + \beta_1 * \text{distance_to_nest} + \beta_2 * \text{colony_size} + \beta_3 * \text{colony_distance} \\ + \beta_4 * \text{Uplift} + \beta_5 * \text{distance_to_dumpsite} + \beta_6 * \text{livestock_prob} \\ + \beta_7 * \text{big_game_hunting_prob} + \beta_8 * \text{roadkill_prob} \\ + \beta_9 * \text{predation_prob} + \beta_{10} * \text{slope} + \beta_{11} * \text{open_areas} + \alpha$$

Where $\beta_1, \dots, \beta_{11}$ are the regression coefficients associated with the explanatory variables, β_0 is the intercept, and α is the random-effect capturing the variability among individuals. Note that the variable distance to the nest was only included in vulnerability and exposure models of adults.

We evaluated the predictive ability of models by calculating the area under the curve (AUC) of a receiver operating characteristic using 80% of the data for training and 20% for testing ([Elith and Leathwick, 2009](#)). We used the “sanity” function to check that the non-linear minimiser showed satisfactory convergence, the Hessian matrix was definite as positive, no extreme or very small eigenvalues were detected, and there were no gradients for the fixed effects larger than 0.001, non-applicable or excessively large fixed-effect standard errors (N/A), sigma parameters smaller than 0.001, and range parameter excessively large.

2.5. Vulnerability and exposure models and collision risk estimation

We projected models obtained for vulnerability and exposure at the scale of peninsular Spain to generate a collision risk map. We first fit a thin plate smoothing spline surface to the variables included in the models using the “Tps” function. We then interpolated them to an empty raster covering the study area at 100 m resolution to predict vulnerability and exposure. As the predicted values for the exposure model were on a logarithmic scale, we converted them into probabilities using the “plogis” function. We estimated collision risk maps for adult and juvenile vultures as the interaction between vulnerability and exposition ([McGlade et al., 2019](#)), cells with values close to 1 indicating a higher collision risk. Finally, we used the “kmeans” clustering algorithm ([Hartigan and Wong, 1979](#)) to classify collision risk into three categories “low risk” (≤ 0.15), “moderate risk” (> 0.15 & < 0.35), and “high risk” (≥ 0.35), and estimate the percentage of area occupied by each category for each age class.

2.6. Collision risk and recorded vulture mortality at wind farms and percentage of breeding population affected

We ran linear models to validate our wind farm collision risk map using field data on recorded griffon vulture mortality at wind turbines in 31 provinces (EU NUTS3 units) and 543 wind farms. Vulture mortality data were obtained by the environmental consultancies that carried out the environmental monitoring plans and were provided to us by the Regional governments. Environmental Impact Assessment carried out by environmental consultancies are usually obliged to transfer bio-monitoring plans to local administrations according to regional regulations (Law 21/2013, of 9 December, on environmental assessment). Wind farms built until 2021 with more than one year of monitoring were

included. We performed this analysis at two spatial scales (province and wind farm) to smooth out differences in monitoring methodologies between wind farms and to make the data comparable. Thus, we calculated an overall estimate of the mortality rate for each province, controlling for the number of operational wind turbines and the year of initiation of operation and mortality monitoring for each wind farm. We obtained a vulture mortality rate per turbine and year in both cases. For the analysis at the province and wind farm level, we used the total number of griffon vulture fatalities without separating by age-classes because this information was rarely available. We then extracted the mean collision risk for each province and wind farm. At the wind farm level, we used a 2 km buffer around each wind farm to obtain the mean collision risk. In order to control for the population size in each province, the number of breeding pairs was added to the model as an offset in the province-level model (Del Moral and Molina, 2018). For both model sets, we estimated how much variability in observed mortality is explained by our model using the “r2_nakagawa” function of the “performance” package (Lüdecke et al., 2021).

We estimated the percentage of the population affected by calculating the mean collision risk within different distance classes. We used the number of breeding pairs within each colony for peninsular Spain from the latest census from Spanish Ornithological Society (see Del Moral and Molina, 2018) and generated 5, 10 and 20 km buffers around each colony point to extract the mean collision risk. Each class distance aims to evaluate whether the risk was higher at small, moderate or large distances from each colony. We then, categorise each collision risk into high, moderate and low (see 2.5 section within methods). Finally, we obtained the percentage of breeding pairs affected within each buffer distance and collision risk category and ran Kruskal-Wallis non-parametric test to assess whether the large percentage of colonies varied within each buffer and collision risk category.

2.7. Current and future estimated impact of wind farms

We overlapped our collision risk map with current operational wind farms and areas where new wind farms will likely be installed. Data on operational wind farms (i.e., georeferenced turbines and energy generated) in 2019, obtained from governments of the Autonomous Communities and completed with the wind energy dataset available online (<https://www.thewindpower.net/>), were used to calculate the number of turbines in each 5 × 5 km grid cell across peninsular Spain.

Wind power density (W/m^2) at 200 m height is useful to identify areas where wind energy could be generated, given the height of new turbine models (Veers et al., 2019). Thus, we extracted it from the 100 m resolution raster format of the Global Wind Atlas (2022) to predict the areas where new wind farms are more likely to be installed in the future. However, since the Spanish legislation strongly discourages the establishment of new wind farms in protected areas, we subtracted these areas (obtained from the protected planet database, UNEP-WCMC, 2023; www.protectedplanet.net) from our wind power density raster.

We generated a 10 km buffer around each cell using the “st_buffer” function of “sf” package (Pebesma, 2018) and extracted the mean number of turbines (current wind farms) and the average wind power density (future wind farms) to assess the impact of current and future wind farms on griffon vultures. We selected this buffer size, double the pixel size, to account for the effects of wind farms in a given cell while reducing pseudoreplication and minimising the overlap between cells. Finally, we ran two linear model sets to assess the effect of the number of operational turbines and wind power density on griffon vultures. The collision risk was entered as a response variable, and the number of turbines or power density was entered as covariates. The response variable was arcsine square root transformed to meet the normality assumption. In addition, the age of the individuals was entered as a factor variable to control for possible differences in current and future turbine exposition for adults and juveniles.

All analyses were performed in R-free software (R Core Team, 2022),

and values are given as mean ± SD. Significance was set at 0.05.

3. Results

3.1. Factors affecting vulnerability and exposure of griffon vultures to wind FARM collision

Overall, our models for vulnerability and exposure showed acceptable predictive capability. The AUC scores for the vulnerability models for adults and juveniles were 0.78 and 0.70, while the AUC for the exposure models was 0.66 and 0.69, respectively (see Figs. S1 and S2). Regarding the specific factors affecting vulnerability and exposure, we found that overall food availability (i.e., anthropogenic carrion sources), the proportion of open areas, the slope and distance to nest and the presence of conspecifics were the main modulators of vulnerability in adults and juveniles (Table 1; Fig. 1A and B). Similarly, we observed that food availability, mainly anthropogenic carrion sources such as big game hunting, and to a lesser extent, flight conditions and distance to nest and presence of conspecifics were related to exposure in adults and juveniles (Table 1; Fig. 1C and D). Estimates of the remaining variables included in the model were not different from 0 (Table 1).

Examining the spatial prediction of our models, they were able to predict well in areas where vulture movement data were available with low uncertainty for both vulnerability and exposure (Fig. 2A and B, Fig. 3A and B). However, as expected, the uncertainty increased outside these areas (Fig. S3A and C), except in the case of juveniles, where the model showed more accurate estimates in northwest Spain (Fig. S3B and D). The final maps of vulnerability and exposure revealed that 19% and 10% of the total Spanish surface is under high collision risk, while areas of 60% and 40% were under moderate risk, and 21% and 50% were under low risk for adults and juveniles respectively (Fig. 2C and Fig. 3C). Significantly, the vulnerability was positively related to exposure, being that relationship more pronounced in the case of adults ($\rho = 0.66$) than for juveniles ($\rho = 0.44$) (Fig. S4).

3.2. Collision risk as a predictor of vulture mortality at wind farms and percentage of affected breeding population

The collision risk predicted with our maps fitted well to the observed mortality rate of griffon vultures at the province level; however, it showed worse performance at the wind farm level (Fig. 4A and B). The observed mortality of griffon vultures was higher in provinces and wind farms with higher collision risk (province: estimate = 2.23, SE = 0.74, t-value = 3.02, $p = 0.005$; wind farm: estimate = 0.825, SE = 0.065, t-value = 12.74, $p \leq 0.001$), explaining 50% and 28% of the total variability observed in the data, respectively.

We found that percentage of population affected differed within each collision risk category (chi-squared = 7.2, $df = 2$, p -value = 0.027) but not in the different buffer distances (chi-squared = 0.09, $df = 2$, p -value = 0.956) (Fig. 5). Notably, our results showed that an important portion of total peninsular breeding pairs ($N = 23,634$) were in areas of high collision risk ($18.3 \pm 0.791\%$), while $37.7 \pm 2.62\%$ and $43.9 \pm 2.9\%$ were at moderate and low collision risk areas, respectively (Fig. 5).

3.3. Collision risk and estimated impact of current and future wind farms

Grid cells (5 × 5 km) with operating turbines were scattered across the whole area, except in central-western Spain, where turbines have not yet been installed (average number of turbines per grid 29 ± 22 ; range = 1–135; Fig. 6A). The areas suitable for new wind farms, based on wind power density and excluding protected areas, covered 66% of the total area of the country (Fig. 6B). The linear models revealed a positive relationship between the collision risk, the number of turbines, and power density. Juveniles showed higher collision risk than adults in areas with more turbines and higher power densities, with a more noticeable effect in the case of power density (Table 2; Figs. 6C and D).

Table 1

Estimates for fixed terms of the full models for vulnerability and exposure for 127 GPS-tagged adults and 50 juvenile griffon vultures in Spain. Abbreviations: SE = Standard Error, Conf.low = Low confidence interval at 95%, Conf.high = High confidence interval at 95%.

	Response	Group	Covariate	Estimate	SE	Conf.low	Conf.high	
Adults	Vulnerability	Conspecific attraction	Nearest colony size	0.007	0.013	-0.019	0.033	
			Distance to the nearest colony (km)	-0.099	0.055	-0.207	0.009	
			Nesting area	Distance to nest (km)	-1.080	0.039	-1.150	-1.000
			Flight conditions	Uplift probability	-0.078	0.074	-0.222	0.066
			Habitat	Proportion of open areas	0.185	0.050	0.088	0.283
			Food availability	Distance to dumpsite (km)	0.006	0.009	-0.011	0.025
		Exposure	Orography	Livestock carrion probability	0.245	0.060	0.129	0.362
				Big game hunting carrion probability	0.686	0.082	0.524	0.847
				Roadkill carrion probability	-0.057	0.043	-0.141	0.027
				Predation carrion probability	-0.019	0.033	-0.085	0.047
				Slope	0.174	0.080	-0.085	0.329
				Slope	0.174	0.080	-0.085	0.329
	Exposure	Conspecific attraction	Nearest colony size	0.017	0.003	0.012	0.022	
			Distance to the nearest colony (km)	-0.247	0.018	-0.283	-0.212	
			Nesting area	Distance to nest (km)	-1.600	0.011	-1.62	-1.580
			Flight conditions	Uplift probability	0.164	0.022	0.122	0.207
			Habitat	Proportion of open areas	0.010	0.013	0.074	0.126
			Food availability	Distance to dumpsite (km)	0.008	0.001	0.005	0.010
		Orography	Livestock carrion probability	0.189	0.014	0.162	0.215	
			Big game hunting carrion probability	0.404	0.023	0.358	0.449	
			Roadkill carrion probability	-0.095	0.016	-0.127	-0.063	
			Predation carrion probability	-0.173	0.015	-0.203	-0.143	
			Slope	0.046	0.021	0.005	0.206	
			Slope	0.046	0.021	0.005	0.206	
Juveniles	Vulnerability	Conspecific attraction	Nearest colony size	-0.012	0.025	-0.061	0.038	
			Distance to the nearest colony (km)	-0.318	0.052	-0.420	-0.216	
			Flight conditions	Uplift probability	0.018	0.083	-0.420	0.181
			Habitat	Proportion of open areas	0.061	0.050	-0.037	0.160
			Food availability	Distance to dumpsite (km)	-0.003	0.018	-0.038	0.033
			Livestock carrion probability	0.085	0.057	-0.026	0.196	
		Exposure	Orography	Big game hunting carrion probability	0.213	0.052	0.110	0.315
				Roadkill carrion probability	0.037	0.045	-0.052	0.125
				Predation carrion probability	0.094	0.054	-0.011	0.200
				Slope	0.161	0.0824	-0.001	0.322
				Slope	0.161	0.0824	-0.001	0.322
				Slope	0.161	0.0824	-0.001	0.322
	Exposure	Conspecific attraction	Nearest colony size	0.035	0.008	0.020	0.050	
			Distance to the nearest colony (km)	-0.103	0.309	-0.163	-0.042	
			Flight conditions	Uplift probability	0.202	0.040	0.123	0.281
			Habitat	Proportion of open areas	0.094	0.019	0.056	0.131
			Food availability	Distance to dumpsite (km)	-0.067	0.004	-0.075	-0.059
			Livestock carrion probability	0.036	0.025	-0.014	0.086	
		Orography	Big game hunting carrion probability	0.308	0.033	0.243	0.373	
			Roadkill carrion probability	-0.152	0.022	-0.195	-0.108	
			Predation carrion probability	0.135	0.034	0.093	0.178	
			Slope	0.212	0.041	0.133	0.291	
			Slope	0.212	0.041	0.133	0.291	
			Slope	0.212	0.041	0.133	0.291	

4. Discussion

Assessing species-specific factors influencing large-scale collision risk and validation with long-term monitoring of fatalities is the gap that needs to be filled to develop evidence-based management actions and minimise the impact of wind energy development on biodiversity as far as possible. In this sense, our work is, to our knowledge, the first large-scale study addressing the factors related to the collision risk of a soaring species such as the griffon vulture using GPS tracking data. This risk is estimated considering its fundamental components –vulnerability and exposure- and the predictive maps obtained are validated with data on vulture mortality observed in turbines. Our results showed that the spatial location of food resources and conspecifics are the main factors affecting vulnerability and exposure in adults and juveniles. Moreover, our prediction of collision risk fits relatively well with observed mortality data, especially when information is pooled at the province level. Noteworthy, ~18% of total Spanish breeding population is within high collision risk areas. Interestingly, collision risk was highest in areas with high turbine numbers and power density, highlighting the need to mitigate conflicts between wildlife and wind energy where turbines are already installed and in areas that could host new turbines in the near future.

4.1. Factors affecting the vulnerability and exposure of griffon vultures to wind farms

Food availability was one of the main factors explaining the vulnerability and exposure of adult and juvenile griffon vultures to wind farms. Despite the unpredictable nature of carrion, griffon vultures prospect at lower altitudes and more frequently in areas where ungulate and livestock abundance is high (Margalida et al., 2011; Cortés-Avizanda et al., 2016; Morant et al., 2022), thus increasing vulnerability and exposure in these cells. Conversely, in areas with less predictable carrions, such as those resulting from predation or roadkill, vulnerability and exposure were low, particularly in adults. Another important predictor of collision risk was the location of nests or breeding colonies. Griffon vultures are highly social and use information from conspecifics in their movement patterns (Cortés-Avizanda et al., 2014; Harel et al., 2017), so cells closer to breeding colonies (or to their breeding site in the case of adults) are more frequently visited by individuals and have a higher exposure value. In these areas close to colonies, birds also reduce their flight altitude, which also increases the vulnerability of a cell. This explains why the spatial distribution and aggregation of individuals are the main predictors of observed mortality in species sensitive to collisions with wind farm turbines, such as griffon vultures (Carrete et al., 2012).

Contrary to de Lucas et al. (2008), who showed high mortality rates of vultures at wind farms located on gentle slopes because of poor wind

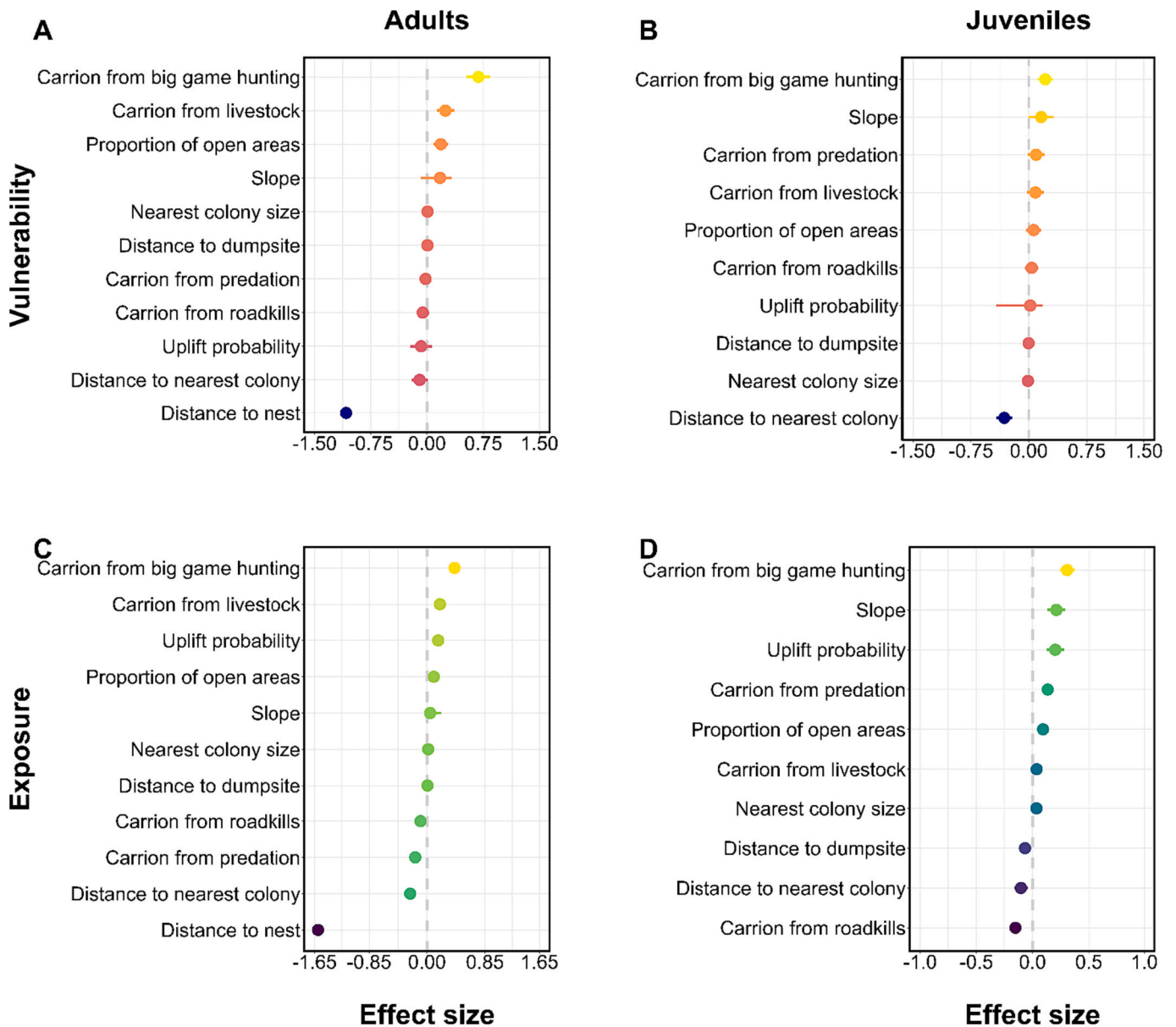


Fig. 1. Estimates for each covariate were included in the best models for Vulnerability (A) and Exposure (B) for adults and juveniles Griffon vultures. The bars showed the 95% Confidence Interval.

conditions, our models predict higher mortalities in cells with higher slopes due to their higher vulnerability and exposure values. Griffon vultures, but also similar species such as condors (Péron et al., 2017), are highly dependent on uplifts for flight and can gain altitude more easily –reducing vulnerability- in open areas where thermal uplifts are more likely, probably due to the higher surface reflectance (see Scacco et al., 2023). Open areas play an essential role in foraging because vultures may find carcasses more easily in these habitats than in closed landscapes (Ruxton and Houston, 2004; Cortés-Avizanda et al., 2014; Oliva-Vidal et al., 2022) but also because livestock carcasses are more frequent in open areas (Arrondo et al., 2019). Although increased visibility and the absence of physical obstacles may improve the detection of livestock carcasses in these habitats and allow vultures to fly at higher altitudes, our results showed that both vulnerability and exposure increased in those areas.

Other factors not considered in this study may also affect vulnerability and exposure of vultures to wind farm collision, in particular, flight type (i.e., soaring vs flapping), manoeuvrability (Péron et al., 2017; Rolek et al., 2022), speed (Masden et al., 2021), visual acuity

(Blary et al., 2023), and avoidance behaviour (Santos et al., 2022). These factors usually operate at a small scale and in limited time windows (e.g., when individuals approach turbines) and could explain the lower ability of our models to predict observed mortality at the turbine scale. Therefore, combining these finer-resolution components with others that act at larger scales could enhance the model capacity to obtain more accurate estimates of collision risk.

4.2. Large-scale map for predicting vulture collision risk at wind farms

Finding links between model predictions and real-world events pose a major challenge in modern ecology (Mouquet et al., 2015). In this context, our work offers evidence for the usefulness of our models by validating them using data from large-scale collisions recorded over 20 years. These validation analyses, which show that the number of observed casualties is positively related to our estimate of collision risk, provide tangible evidence of the usefulness of this model beyond its inherent statistical performance (Wadoux et al., 2021). To our knowledge, this work is the first to compare expected collision risk with

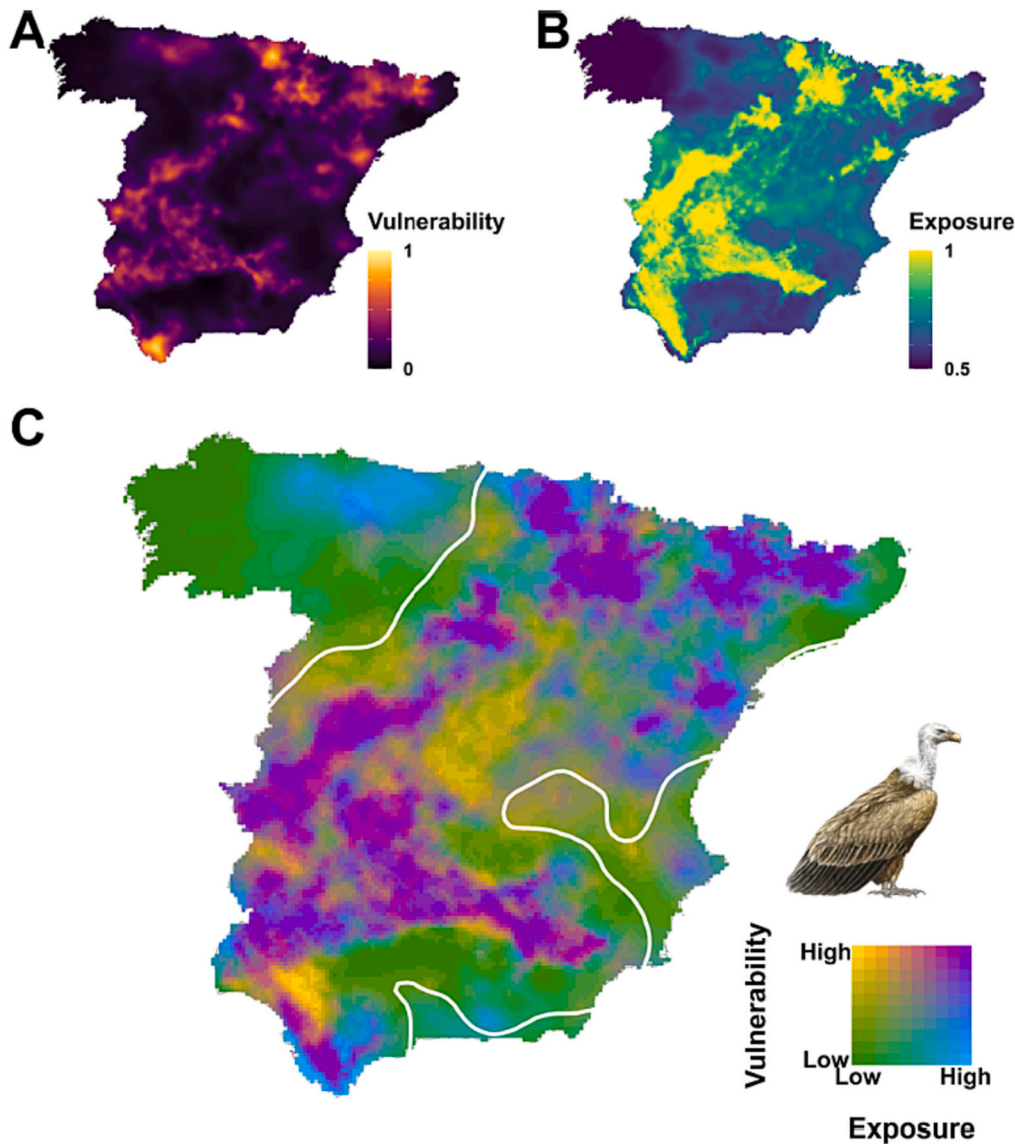


Fig. 2. Predicted probability extracted from the best model of Vulnerability (A) and Exposure (B) for adults. Note that predicted values for exposure were transformed into probabilities using the “plugins” function. Fig. C represents a bivariate raster generated from the predicted probabilities of vulnerability and exposure to identify areas of collision risk at 5 km resolution for Peninsular Spain. The white contour represents the KDE 99 estimated from the flight locations of 127 GPS-tagged adult Griffon vultures. Adult and juvenile griffon vulture sketches by Rafael A. Galvez.

observed bird mortality data, representing a step forward in the evaluation of collision risk maps derived from GPS data or other data sources (e.g., Thaxter et al., 2019; Murgatroyd et al., 2021; Gauld et al., 2022; Cervantes et al., 2023). However, a crucial point to note is the low congruence between the results obtained in our maps and the observed mortality at the wind farm level, especially in cases where the observed mortality rates are high but the predicted collision risk is low. This result shows the reduced performance of our collision risk map at small scales, mainly due to other untested factors such as the presence of unknown supplementary feeding sites nearby of some wind farms or differences between wind farms in mortality reporting standards, technical characteristics (e.g., turbine type) or mitigation measures implemented (e.g. Ferrer et al., 2022; McClure et al., 2021). Finally, it is worth noting that another important aspect that may be affecting the performance of the collision map at small scales relates to the character of the data. Collisions are rare and discrete single events, (although their sum could be high at large spatio-temporal scales), that occur in a large number of turbines. Thus, detecting effects at the turbine level would require much more data than at the spatially integrated level of a region. This

highlights the need to collect more data within wind-farms to detect turbine-level effects and implement more efficient and effective corrective measures.

It is also important to consider that our results, particularly the map of exposure, could be influenced by the origin of the information (certain populations studied), which could influence the results obtained. Thus, despite their general usefulness for large-scale wind energy planning, our collision risks maps cannot be used blindly without considering specific aspects of the different areas and characteristics of the wind farms to be installed. On the contrary, maps should be applied as a guideline to rule out risky areas or direct monitoring and mitigation efforts. In this sense, our models show that almost 20% of the Spanish breeding population of griffon vultures is located in areas of high collision risk, where measures to reduce this unnatural mortality and monitoring programs should be considered a priority. In areas where medium-low risk is predicted, however, the precautionary principle should be applied and quality environmental impact studies should be required to minimise potential future conflicts as these areas agglutinate the remaining 80% of the breeding population of the species.

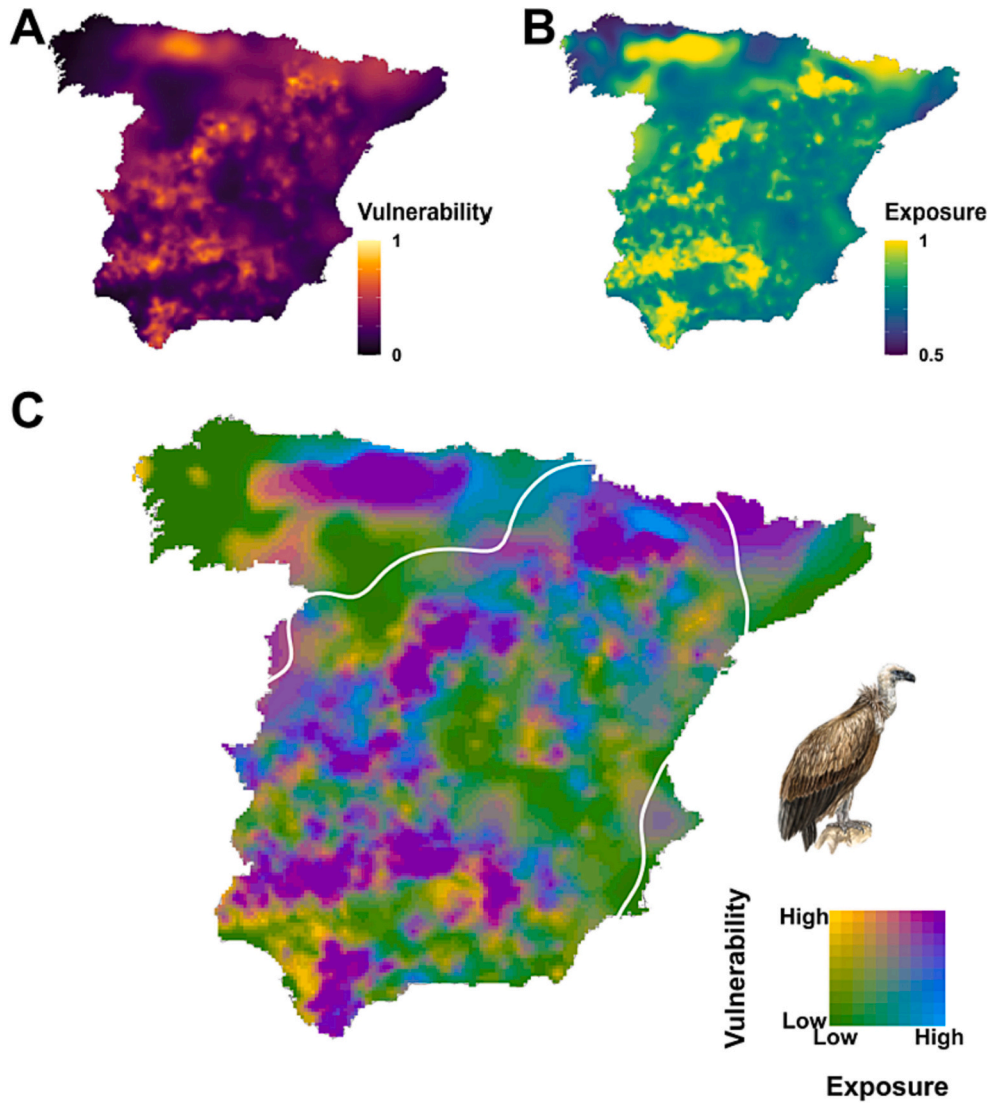


Fig. 3. Predicted probability extracted from the best model of Vulnerability (A) and Exposure (B) for juveniles. Note that predicted values for exposure were transformed into probabilities using the “plogis” function. Fig. C represents a bivariate raster generated from the predicted probabilities of vulnerability and exposure to identify areas of collision risk at 5 km resolution for Peninsular Spain. The white contour represents the KDE 99% estimated from the flight locations of 50 GPS-tagged juvenile Griffon vultures.

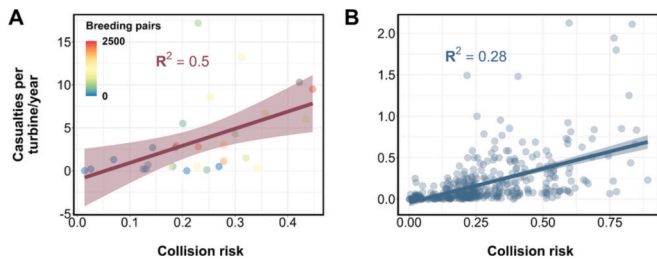


Fig. 4. Relationship between the rate of griffon vulture casualties in 31 Spanish provinces for all the birds and mean collision risk per province (A) and at wind farm level (B). The R^2 represent the variability explained by the linear regression model. In Fig. A, points represent raw data corresponding to each province, while colour intensity corresponds to the number of breeding pairs. In Fig. B, raw points correspond to each wind farm.

Given the current and future wind energy development and the surface that our models classify as high collision risk for griffon vultures (ca. 26% and 58% for adults and juveniles, respectively), efforts to

identify areas and strategies to minimise this conservation conflict are essential. For example, our analyses show that areas with a large number of turbines present a high collision risk and therefore, wind farms planned in the vicinity of existing wind farms should be assessed with special attention. Concerning this, the strategy of fragmenting future wind farm projects into several small ones to reduce their individual impacts should be combated by the responsible administrations by analysing the location of the project within the overall map of existing and planned wind farms. In addition, it is highly recommended to enforce measures that reduce mortality rates in existing wind farms, such as in situ turbine stopping protocols (Ferrer et al., 2022). Other systems, such as automated curtailment, have shown interesting results in reducing collisions in raptors from North America (McClure et al., 2021). However, their effectiveness should be evaluated before promoting their deployment in other species and locations.

Assuming a high-power density is incorporated into new wind farms, our results suggest that the impact would be even greater than today, particularly in areas of high collision risk and power density. This scenario is highly plausible (but see Kemfert et al., 2022) given the global “green energy wave” being undertaken around the world and, in

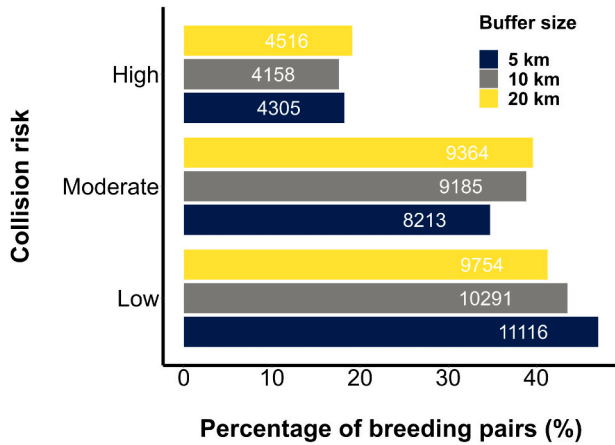


Fig. 5. Percentage of breeding pairs at high, moderate and low collision risk within 5, 10 and 20 km buffer around colonies. The numbers in white represent the total number of breeding pairs within each category.

particular, Spain, which is responsible for 60% (10,378 GWh) of the total wind energy generated worldwide (Katzner et al., 2019; Serrano et al., 2020; Dunnett et al., 2022). This unprecedented scenario jeopardises biodiversity in general and, particularly, highly vulnerable taxa such as raptors and bats (Serrano et al., 2020; Pérez-García et al., 2022). Given that the number of vulture casualties at wind farms is of concern

both globally and locally (Santangeli et al., 2019; Pérez-García and Serrano, 2023), it is necessary to prioritise areas based on species behaviour to better inform potential risk areas both in the short and long term, similarly to how it was done to identify hotspots of bird mortality by electrocution (e.g. Pérez-García et al., 2017). In this context, our work is a step further towards obtaining an accurate spatial risk assessment of vulnerable species to help prioritise areas where current and future wind farm development should be avoided. The maps from this research would be the baseline for identifying “not go-to” areas for renewable energy deployment in Europe, as required by the actually proposed Directive 2022/0160/EU (COD), as part of the Repower EU plan to promote renewable energy infrastructure.

Our modelling shows that large areas present a low risk of collision for griffon vultures (ca. 21% and 50% of Spanish surface for adults and juveniles, respectively). As mentioned above, many of these areas show

Table 2

Results for the linear models for the current and future impact of wind turbines on collision risk. Abbreviations: SE = Standard Error.

	Terms	Estimate	SE	t-value	P-value	R ²
Current turbine impact	No. of turbines	0.011	0.002	4.639	<0.001	15.26
	Age (Juveniles)	0.060	0.005	13.216	<0.001	
Future turbine impact	Power density	0.009	0.002	10.25	<0.001	14.53
	Age (Juveniles)	0.140	0.002	81.32	<0.001	

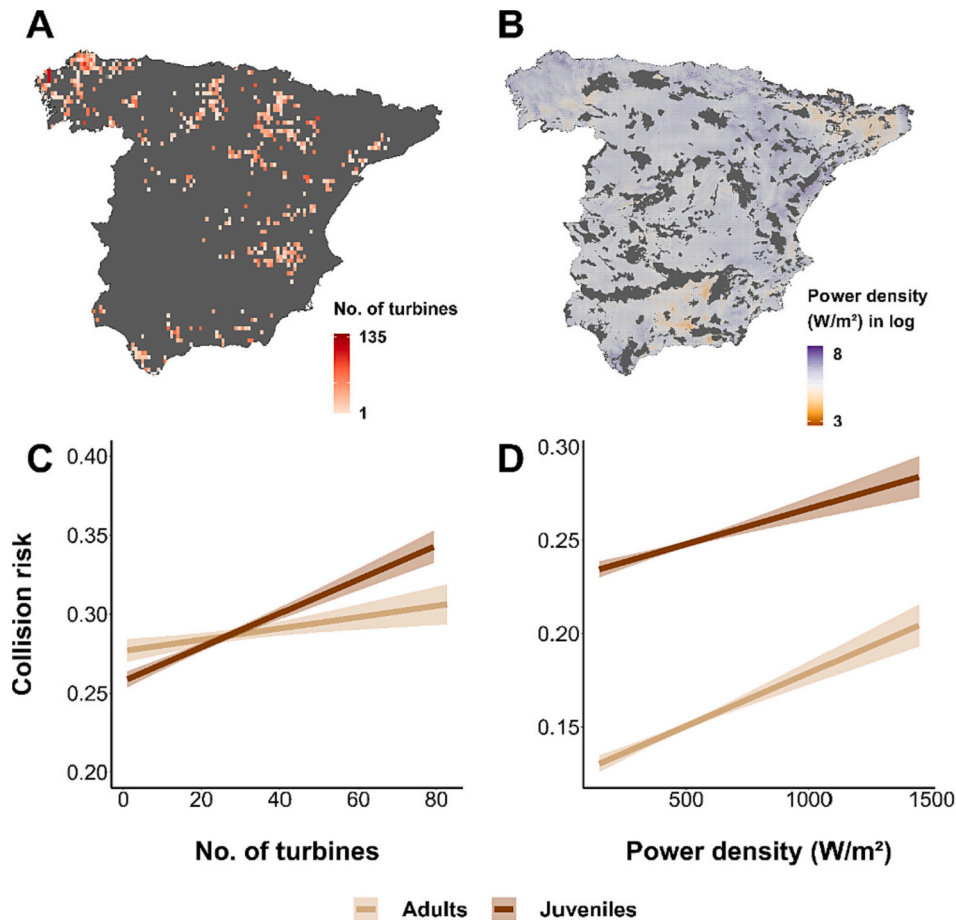


Fig. 6. The number of wind turbines present (A) (see Turbine data extraction in the method section for details) and power density at 200 m (i.e., the estimated height for large turbines) in peninsular Spain (B). Figs. C and D represent the average number of turbines and power density (w/m²) respecting collision risk for adults, juveniles and the average. Figs. A and B presented data in raster format in 5 × 5 km grid cells. No data values are represented in dark grey colour.

high values of observed vulture mortality, so the precautionary principle should always be used before approving any wind farm there. It is also important to consider that map resolution could also affect the percentage of the affected population; for instance, it may underestimate the number of breeding pairs within high or moderate-risk areas. Furthermore, it is essential to remember that these areas have only been projected using a species, the griffon vulture, studied in a specific period. Thus, those areas could pose serious collision risks for other species for which GPS data does not exist but show high collision rates at turbines (e.g. short-toed snake-eagle *Circaetus gallicus*; Pérez-García and Serrano, 2023). Moreover, these areas may also harbour species that may be affected by wind farms indirectly, mainly through habitat disturbances or transformation (Gómez-Catasús et al., 2022) or collision with associated infrastructures such as power lines (Arrondo et al., 2020; Marcelino et al., 2018). Integrating GPS information of other species affected by wind farms might reveal new risky areas and could help refine the selection of suitable and unsuitable areas for the deployment of new energy facilities (i.e., wind farms or photovoltaic plants) (see, for example, Gauld et al., 2022). This interspecific approach may alleviate the conflict and reconcile wildlife and energy generation by using evidence-guided spatial planning.

5. Conclusion

Our work evidenced the usefulness and validity of combining vulnerability and exposure to estimate and predict the collision risk of a soaring species in wind farms. For griffon vultures, the installation of wind farms should avoid areas where there are breeding colonies of the species, as well as areas used for extensive livestock farming and where ungulate populations are high, as these areas are the most heavily prospect by vultures at relatively low flight heights. This supports the hypothesis that the spatial distribution and aggregation of species vulnerable to collide with turbines should be used as criteria for large-scale environmental planning. The latter is also supported by the non-negligible percentage of total Spanish colonies located in high risk areas. Furthermore, given the good performance of the griffon vulture as a bioindicator of turbine mortality for other flying species, the risk maps obtained in this study could be used as an approximate baseline of unsuitable areas for species for which no data are available. Relevantly, our study highlights the need to promote and support long-term standardised mortality monitoring at wind turbines to better assess the validity of coarse and fine-scale spatial predictions. However, it should not be forgotten that the development of renewable energies entails multiple impacts that can affect several species differently, so it is essential to advance in multi-species predictive models. Moreover, developing and combining similar approaches to include the impacts of other renewable energy sources, such as solar farms, would be desirable. As the demand for renewable energy rises and is expected to double its production by 2030 (IRENA, 2020; EU Commission, 2021), urgent measures are needed to find alternatives with low biodiversity impacts and mitigate emerging conflicts between human-made infrastructures and biodiversity conservation.

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Ethics statement

All procedures regarding animal manipulation and tagging were carried out strictly in accordance with the relevant guidelines and regulations of the “Patrimonio natural y de la Biodiversidad” (Article 61, Law 42/2007, Ministerio para la Transición Ecológica y el Reto Demográfico, España) and complied with all regional permits (regional government of Castilla y León, Junta de Andalucía, Comunidad Foral de Navarra, Generalitat de Catalunya, Gobierno de Aragón and Diputación Foral de Gipuzkoa).

Credit authorship contribution statement

JM Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Software, Validation, Visualization, Writing-original draft. JMPG Conceptualization, Data curation, Funding acquisition, Investigation, Project administration, Resources, Supervision, Validation, Writing-review & editing EA, JASZ, JAD, MC, FG, DS and JMPG Supervision, Writing-review-editing. All authors gave final approval for publication.

Declaration of Competing Interest

The authors have no conflicts of interest to declare.

Data availability

The data that support the findings of this study are openly available in Dryad at <https://doi.org/10.5061/dryad.fxpnvx0xt>

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.eiar.2023.107339>.

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