Environmental Impact Assessment Review 110 (2025) 107669

Contents lists available at ScienceDirect

Environmental Impact Assessment Review

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

Cumulative collision risk and population-level consequences of industrial wind-power plant development for two vulture species: A quantitative warning

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ARTICLE INFO

Keywords:
Collision mortality
Integrated population models
Spatial planning
Wildlife management
Wind energy

ABSTRACT

Prioritizing renewable energy generation over the conservation of natural habitats and species on a large spatial scale, leads to the paradox of impacting biodiversity to mitigate climate change. In this study, we aim at quantifying the long-term demographic impact of the excess mortality caused by collisions with wind turbines on the populations of two vulture species of conservation concern. Using long-term monitoring data and Integrated Population Models (IPMs), we quantified demographic parameters and projected population trends under various wind power development scenarios. Our findings indicate that even under our most optimistic scenarios, annual collision mortality could reach up to 30 % of the current Cinereous vulture population and 7 % of the Griffon vulture population. Without further wind power expansion, both vulture populations are predicted to remain stable or increase over the next 20 years. However, the addition of 85 wind turbines is likely to drive the Cinereous vulture to local extinction within 18 years and significantly slow the growth of the Griffon vulture population. Scenarios involving larger numbers of turbines could result in the extinction of both species within two to five years for Cinereous vultures and up to 20 years for Griffon vultures, depending on space use intensity. Our results underscore the vulnerability of long-lived species to excess mortality and highlight the need for comprehensive Environmental Impact Assessments (EIAs) that incorporate population dynamics analyses. Effective conservation strategies must include rigorous pre- and post-construction monitoring, the availability of monitoring data, and cumulative impact assessments that consider the entire foraging range of these species. Additionally, strategic planning to avoid critical vulture habitats and implementing mitigation measures in buffer zones are essential. This study emphasizes the necessity of integrating biodiversity considerations into renewable energy planning to balance the goals of energy production and wildlife conservation.

1. Introduction

Harnessing wind power for electricity production through onshore wind power plants, plays a significant role in the transition to renewable energy and the mitigation of climate change. Over the past decade, wind power generation has experienced substantial global expansion, with a notable increase of almost 30 % (IRENA, 2020). Nevertheless, this method of energy production sometimes has negative impacts on biodiversity, and can be a severe cause of mortality, particularly for birds (Drewitt and Langston, 2006; Watson et al., 2018; Serrano et al.,

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https://doi.org/10.1016/j.eiar.2024.107669
Received 9 June 2024; Received in revised form 9 September 2024; Accepted 9 September 2024
Available online 13 September 2024

2020). Wind power plants can impact birds either directly by collision with turbines or indirectly as a result of displacement, habitat loss/degradation and increased energy expenditure, acting as barriers to their regular flight paths (Marques et al., 2014; Schuster et al., 2015; Jager et al., 2021). The vulnerability of large raptors to collision-induced mortality can be attributed to their large spatial requirements and their utilizing of ridges, where updrafts are strong, for their daily movements (Morant et al., 2023; Tobajas et al., 2024), their narrow field of view (Martin et al., 2012), as well as other morphological and behavioural characteristics that render them particularly vulnerable when flying in the vicinity of operating wind turbines (Thaxter et al., 2017). From a demographic perspective, their long lifespan, delayed sexual maturity and low reproductive rates make populations highly sensitive to any additional mortality, even at a level of a few individuals per year over a long period (Whitfield et al., 2004; Duriez et al., 2023). Additionally, young vultures undergo an extended natal dispersal phase (Penteriani et al., 2011; Serrano, 2018) during which they cover large areas and face various risks (Oppel et al., 2015; Buechley et al., 2021; García-Macía et al., 2023; Martínez et al., 2024).

Prioritizing the generation of clean energy over the preservation of natural habitats and species on a large spatial scale, leads to the paradox of exacerbating biodiversity loss to mitigate climate change. This study focuses on the effect of wind power plants built in the Greek part of the trans-border Eastern Rhodopes mountains in south-eastern Europe. The area encompasses five Special Protection Areas (SPAs) within the Natura 2000 Network, including the Dadia-Lefkimi-Soufli Forest National Park. The area supports the only native Cinereous vulture (*Aegypius monachus*) population in the Balkans, comprising of 30–36 breeding pairs (overall around 120 individuals), that exclusively nest within the Dadia-Lefkimi-Soufli Forest National Park. While the breeding population has been stable or slightly increasing during the last three decades, it still experiences considerable mortality mainly due to poisoning (Skartsi et al., 2014) and collision with wind turbines (Vasilakis et al., 2016; Vasilakis et al., 2017), while a large-scale wildfire in 2023 led to the deterioration of a significant portion of its breeding habitat. The Cinereous vulture uses a large foraging area (Vasilakis et al., 2008), covering considerable distances from the breeding colony, even reaching former nesting sites in Bulgaria (Arkumarev et al., 2020). The species is classified as Endangered (EN) in Greece and globally as Near-Threatened (NT). The Eastern Rhodopes also host a large (380 individuals) Eurasian Griffon vulture cross-border population that nests, roosts and forages in and around the wind power plant development area. The population consists of a few pairs that nest in the Greek part of the region whereas the largest breeding colony is located in the Bulgarian part (Demerdzhiev et al., 2014; Dobrev et al., 2022). Griffon vultures from those colonies frequently undertake large foraging trips within the study area (Arkumarev et al., 2021). The species is classified as Critically Endangered (CR) in mainland Greece and globally as Least Concern (LC) with a positive population trend, exhibiting varying population dynamics across countries (BirdLife International, 2024).

Despite its profound importance for raptors and vultures, a substantial part of the region was designated as one of the highest priority areas for wind power plant development by the Greek government in the current Special Framework for Spatial Planning, based on a Strategic Environmental Assessment (Kafetzis et al., 2017). Hundreds of wind turbines have been already installed in the area and applications for the installation of even more are underway. Until mid-2010s most (84 %) of the 185 operating turbines (253 MW) were installed within the most significant zone for the conservation of the Cinereous vulture (Vasilakis et al., 2016).

Evaluating the potential risks that any excess mortality, induced by any plan or project, may pose to protected species' populations is an obligatory procedure through the implementation of an Environmental Impact Assessment (EIA), or in EU countries under Article 6.3 of the Habitats Directive, within the process of Appropriate Assessment conducted prior to granting the relevant permits. Particularly for windfarms, it is typical for both EIAs and post-construction mortality surveys to focus solely on quantifying the number of animals killed or at risk of being killed annually. In fact, risk assessment methods used in EIAs have proven to be inadequate as they often assume a linear relationship between the frequency of observed birds and fatalities, which does not hold true in practice (Ferrer et al., 2012). Despite the fact that the potential impact of added mortality on a population's dynamics is often the objective of several research studies, it is seldom quantified in EIAs, especially considering the cumulative impacts of all wind power plants in a region (Masden et al., 2010; Green et al., 2016). In addition to assessing fatality risk, it is therefore necessary, to move from the individual to the population scale; the comprehensive quantification of population-scale impacts should be the central concern of EIAs and Appropriate Assessments to effectively determine whether the targeted infrastructure project, on its own or in synergy with existing ones, poses a threat to the conservation status of a specific set of species. Such an approach will not only allow to inform management decisions to mitigate threats but to also highlight the severity of the issue to politicians, stakeholders and the general public (May et al., 2019).

In this study, we aim at quantifying the long-term demographic impact of the excess mortality caused by collisions with wind turbines on the populations of two vulture species of conservation concern. We first estimate the cumulative current and future collision mortality based on the installed and planned wind power plants in the study area. Then, we assess the collision mortality effects at the population scale by calculating demography and extinction probability at three future scenarios of wind energy development intensity. Finally, we assess results in relation to conservation planning for the designated sites and populations, providing a critical view on the species' persistence.

2. Material and methods

2.1. Collision risk assessment

2.1.1. Movement data

Between 2016 and 2022 we tagged 40 Cinereous vultures (9 Juveniles, 14 Immatures, 4 Subadults, 13 Adults) and 32 Griffon vultures (13 Juveniles, 6 Immatures, 3 Subadults, 10 Adults), representing 33 % and 9 % of the estimated species populations for 2016–2022, respectively (Table S1; Table S2). Both species were trapped during the non-breeding season using walk-in cages with sliding doors. For Cinereous vultures, trapping occurred in September–January, while Griffon vultures were trapped in June–October. Individuals were marked with plastic and metal alphanumeric rings and equipped with solar-powered Argos/GPS or GPS/GSM transmitters (Ecotone https://ecotone-telemetry.com/en, e-Obs https://e-obs.de/, Argos https://www.argos-system.org/). Transmitters were attached using backpack or leg-loop harness configurations with 11.2 mm Teflon ribbon (Anderson et al., 2020), ensuring that the weight of transmitter harness, rings, and wing tags did not exceed 3 % of the bird's body mass, to avoid any adverse effects. The age of individuals was estimated from plumage traits (Clark, 2004; De la Puente and Elorriaga, 2004; Zuberogoitia et al., 2013).

The transmitters were programmed to record GPS positions from sunrise to sunset at various intervals (3–120 min) due to varying technologies and sensors applied (Table S1). All data were stored in and accessed through Movebank (www.movebank.org; (Kranstauber et al., 2011). Prior to analyses, the data were inspected and visualized to check for outliers. Using the Movebank data filters and manual exclusions of any remaining extreme coordinates, we removed erroneous GPS fixes (Walter et al., 2011) and clipped movements out of the study area.

2.1.2. Wind-power plant data

From the 317 wind power plant projects encompassing 1284 turbines reported for the area in the publicly available geoportal of the Regulatory Authority for Energy, Waste and Water (RAEWW; https://geo.rae.gr/), we identified distinct operational stages for each one, including producer's certification, environmental approval, installation permit, and operation permit. As of March 2022, 26 wind power plants with 265 turbines were operational, 2 wind power plants (11 turbines) had installation permits, and 11 wind power plants (74 turbines) had acquired environmental licensing, totalling 350 wind turbines (Current Wind Power Plants, CWP; Table 1). Additionally, the future scenario involves 278 wind power plants with producer's certification, totalling 934 wind turbines (Future Wind Power Plants, FWP; Table 1). Notably, more than 61 % of CWP (corresponding to 51.6 % of the installed power in MW) were located within Special Protected Areas (SPAs). For FWP, more than 33 % of turbines and 32 % of power are planned within SPAs.

Detailed technical characteristics, including hub height, rotor diameter, etc., for all 317 wind power plants (1284 turbines, both CWP and FWP) were also obtained from RAEWW. Information on the maximum chord, pitch, and rotation period (fastest) of each turbine model was sourced from EIAs for each wind power plant. In cases of variable pitch, a conservative assumption of 5 degrees was adopted (maximum pitch). Turbine operation time was assumed to encompass daylight hours throughout the year.

2.1.3. *Data treatment and analyses*

The tracking dataset was used to estimate flying speed and height over ridges, the proportion of the population using each wind power plant, and the population Utilization Distribution (UD) estimations for both the breeding season (Cinereous vulture: February–August; Griffon vulture: January–July) and the non-breeding season (Cinereous vulture: September–January; Griffon vulture: August–December). For the Cinereous vulture UD estimation, the 2016–2022 dataset was subsampled at least at hourly intervals, retaining only the fixes closest to each hour change to achieve independence. Only individuals with over 50 locations per season were included in the analysis.

Breeders were expected to spend an increased amount of time in and/or around the nest, and to prevent location aggregation around nesting sites we randomly selected one location per tracking day from the set of locations inside a 50 m buffer around nests, from the incubation onset period until failure or one month post hatching (D'EON and Delparte D'eon and Delparte, 2005; Vasilakis et al., 2016). We included all locations up to the home range stabilization per individual and season, determined by plotting home range size versus the number of locations (Kenward, 2000). Then we employed Fixed Kernel (FK) home range analysis (Kie et al., 2010) using a grid cell of 200 × 200 m and an iterative plug-in bandwidth selection (Amstrup et al., 2004; Gitzen et al., 2006) in order to estimate individual Utilization Distributions (UDs) for each season. Next, we estimated the seasonal UD for the average individual by summing up the seasonal individual UDs from all the vultures in the sample and we divided with the magnitude of the sample (n of individuals). Finally, we multiplied the latter seasonal UD of the average individual with the estimated size of the population to estimate seasonal population UD. Each cell of this UD gives the percentage of time spent by the population in the pixel. Data processing and estimations were completed in R v4.0.2 (R Core Team 2020) using packages 'plugdensity' (Herrmann and Maechler, 2023) and adehabitatHR (Calenge, 2019).

We used site-specific GPS-derived flight speed, as generic values can bias collision risk estimates (Masden et al., 2021). Flight height and speed were estimated by isolating last fixes from bursts, where speed was >4 m/s (to ensure that only flight data were included). We further filtered those fixes that were within 200 m of ridges to obtain the subset of vulture behavior close to sites where wind power plant establishment is more likely. A detailed description of flight height and speed estimation can be found in the Supplementary material.

2.1.4. *Collision rate estimations*

We integrated the UD of each species, obtained through the FK method, with a "Band" Collision Risk Model (CRM; Band et al., 2007), to assess potential collision mortality. The annual mortality for the CWP was determined by aggregating seasonal mortalities. This involved calculating the percentage of time spent by the vulture population per km^2 for each pixel, per season (t_S), adjusting for pixel area (0.04 km^2). Predicted seasonal mortality for each wind power plant was derived by incorporating specific parameters into the CRM: (a) percentage of time spent by the population fraction using each wind power plant area (200 m buffer around the turbines); (b) total vulture activity duration during breeding and non-breeding seasons; (c) percentage of time flying at collision risk height (Table S3; Table S4); (d) the species' morphological and estimated flight parameters (WAZA 2014; Cinereous vulture: length = 1.11 m, wingspan = 2.75 m, flying speed = 11.9 m/s; Griffon vulture: length = 1.11 m, wingspan = 2.53 m, flying speed = 13.23 m/s), and (e) technical characteristics of the CWP (Table S5). Analysis was conducted for five avoidance rates (95 %, 98 %, 99 %, 99.5 %, and 99.9 %), given that the avoidance rate is the most influential component of the Band CRM's outputs (Chamberlain et al., 2006).

Annual collision mortalities for FWP were estimated using the same approach as in CWP. To determine a representative vulture population fraction utilizing the FWP in the study area, we employed three scenarios: a) the maximum observed use from CWP (Cinereous vulture: 83 % and 63 % of the population for breeding and non-breeding season respectively; Griffon vulture: 63 % and 60 % of the population for breeding and non-breeding season respectively), b) a medium, arbitrarily set use at 50 % for both species, and c) the average observed use from CWP (Cinereous vulture: 34 % and 29 % of the population for breeding and non-breeding season respectively; Griffon vulture: 26 % and 25 % of the population for breeding and non-breeding season respectively). Finally, to estimate the percentage of time that vultures fly at collision risk height of the FWP we used the average rotor swept heights of operating wind turbines and the frequency distribution of flight heights over ridges for both species.

Table 1
Wind turbine numbers at different operational stages within and out of Special Protected Areas (SPAs) sites in our study area. Numbers in brackets indicate produced power in megawatts.

	Wind turbine operational status	Within SPA	Out of SPA	Total
Current Wind Power Plants (n = 39)	Operating	173 (257.4)	92 (206.8)	265 (464.2)
	Permission of installation	11 (44.4)	0 (0)	11 (44.4)
	Environmental licensing	30 (95)	44 (163.7)	74 (258.7)
	Sub-total	214 (396.8)	136 (370.5)	350 (767.3)
Future Wind Power Plants (n = 278)	Permission of production	309 (1237.2)	625 (2643.9)	934 (3881)
	Total	523 (1634)	761 (3014.4)	1284 (4648.3)

2.2. *Estimation of demographic parameters in an Integrated Population Model*

2.2.1. *Population monitoring*

An annual monitoring scheme of the Cinereous vulture breeding activity in Dadia-Lefkimi-Soufli Forest National Park was initiated in 1987 and is carried out until today, considered one of the few long-term monitoring schemes of birds in Greece (Skartsi et al., 2010). Each year, the nests are monitored from February to August, in four stages, during which nest occupancy, incubation, hatching and fledging are recorded during systematic monthly visits (Poirazidis et al., 2007). In this study we included long-term data of breeding population counts (number of breeding pairs), and breeding output data (productivity and breeding success of monitored nests) for 1993–2023. The monitoring of the Griffon vulture breeding population both in Bulgaria and Greece was implemented annually during January – August, with at least three visits: the first visit was conducted in February to record the number of incubating pairs and occupied nests. The second visit was in May to establish the breeding success of the pairs. The third visit was in July to record the number of juveniles in the nests before fledging (Demerdzhiev et al., 2014). Similarly to the Cinereous vulture, in this study we used long-term data of breeding population counts and breeding output for 1989–2023.

2.2.2. Model implementation

We constructed two Integrated Population Model (IPM) parameterised for each species to combine the different datasets and account for uncertainty in demographic parameters in population projections (Fig. 2). For both species, we built pre-breeding age-structured female-based matrix models with six different age classes inspired by other demographic studies on vulture species (Lieury et al., 2015; Margalida et al., 2020; Oppel et al., 2021). The models assume an equal sex ratio at hatching and that all adult birds (> 5 cy) attempt to reproduce every year. Individuals older than 1 year and before reaching 4th year were assumed to have the same annual survival rate (S_{imm}) as immatures, birds that are older than 4 years until reaching adulthood have an annual survival rate S_{sub} for subadults, birds aged <1 year old have an annual juvenile survival (S_j) and the annual survival of birds that are over 5 years old is defined as adult survival (S_{ad}). Although there could be more age structure in the survival of long-lived species such as the Cinerous and Griffon vultures, we followed the age class delimitation implemented in other modelling studies for the species (Van Beest et al., 2008; Rousteau et al., 2022). The resulting life-cycle graph is shown in Fig. S1.

We did not assume demographic stochasticity in the population model, but rather used a population growth process using the Leslie matrix method. However, we included environmental stochasticity both in productivity and survival estimates. Although immigration can be an important driver of population dynamics in small populations, we did not include an immigration component in our population model. The Cinereous vulture population in Dadia NP seems to be geographically isolated (Vasilakis et al., 2008; Vasilakis et al., 2016) and is located >400 km from the nearest native source population that could provide a number of immigrants (Arslan and Kirazli, 2022). Only four such movements have been recorded during the last three decades, with two of them regarding disoriented juveniles (WWF Greece and NECCA, unpublished data), therefore the number of immigrants is likely a negligible source of the population trajectory in the studied population so far. Although a reintroduction effort of the species is currently implemented in Bulgaria, the programme is too recent (after 2020), to expect any important influence. Furthermore, IPMs could overestimate the contribution of immigration in population growth changes especially since we lack any evidence of variation and any explicit data on immigration (Paquet et al., 2021).

Briefly, we used an hierarchical state-space model to describe the population trajectory of each species (Cinereous vultures between 1993 and 2022; Griffon vultures between 1989 and 2022). The observation process, i.e., the relationship between the annual census data of breeding pairs and true population size, was modeled using a Poisson distribution, a Poisson regression model was used to analyse productivity data (annual total number of fledglings produced by the surveyed broods) and a known-fate model was fitted on capture histories of the GPS-tracked individuals. A detailed model description along with component likelihoods and set priors can be found in the Supplementary material.

We fitted the integrated population model in JAGS (Plummer, 2012) called from R 4.2.0 (R Core Team 2022) via the package 'jagsUI' (Kellner, 2015). We ran three Markov chains each with 10,000 iterations and discarded the first 2000 iterations. We tested for convergence using the Gelman-Rubin diagnostic (Brooks and Gelman, 1998) and confirmed that R-hat was <1.01 for all parameters. We present posterior estimates of parameters with 95 % credible intervals.

2.3. Population projections under different scenarios of wind power plant development

To examine whether future establishment of wind power plants would affect the studied vulture populations we projected the population size estimated by the integrated population model 20 years into the future while accounting for the uncertainty in demographic parameters (Schaub and Abadi, 2011; Oppel et al., 2014). We used the mean survival probabilities for each age class and the mean productivity to project population growth into the future, and incorporated different scenarios of wind energy development, according to section 2.1.2.: a) operation of an additional 85 wind turbines, 11 of them with installation permits and 74 having acquired environmental licensing (CWP; Table 1), under a 98 % avoidance rate scenario, leading to 1.94 dead Cinereous vultures and 3.76 dead Griffon vultures per year (AEPO scenario; Table S3, Table S4); b) future establishment of 934 wind turbines (FWP; Table 1), using the average observed use from CWP (Cinereous vulture: 34 % and 29 % of the population for breeding and non-breeding season respectively; Griffon vulture: 26 % and 25 % of the population for breeding and non-breeding season respectively), leading to 28 dead Cinereous vultures and 12 dead Griffon vultures per year under a 98 % avoidance rate scenario (FWP-average use scenario; Table S3, Table S4); c) future establishment of 934 wind turbines (FWP), that if used by 50 % of the population for both species, would lead to 44 dead Cinereous vultures and 21 dead Griffon vultures per year under a 98 % avoidance rate scenario (FWP-half pop use scenario; Table S3, Table S4); d) future establishment of 934 wind turbines (FWP; Table 1), using the maximum observed use from CWP (Cinereous vulture: 83 % and 63 % of the population for breeding and non-breeding season respectively; Griffon vulture: 63 % and 60 % of the population for breeding and non-breeding season respectively), leading to 65 dead Cinereous vultures and 28 dead Griffon vultures per year under a 98 % avoidance rate scenario (FWP-maximum use scenario; Table S3, Table S4). In each scenario, mortality estimates were distributed across all age classes and for the entire population. We run the IPM scenarios assuming a 98 % avoidance rate, since it is generally recommended for vultures (Vasilakis et al., 2016) and additionally it was already proposed for the Cinereous vulture according to carcass survey results (SNH, 2018; Vasilakis et al., 2016). Furthermore, it is also widely accepted by assessors and evaluators and has been regularly used in pre-construction Appropriate Assessments for most of the wind power plants in our study area.

3. Results

3.1. Current and future collision mortality

The UDs for both vulture species were found to converge at specific high-use SPAs in Greece used for breeding and/or foraging (Fig. 1). Cinereous vultures spent on average 21 % and 24 % of their time (during the breeding and non-breeding season respectively) flying at the rotor risk heights, and 28 % of their overall time flying within the 200 m buffer zone around the CWP, covering an area of 32.78 km^2 (Table S3, Table S5). We predicted an overall annual Cinereous Vulture collision mortality for the CWP of eight individuals under a 98 % avoidance rate (Fig. 3; Table S3), whereas it varied from two up to 20 deaths under 99.5 % to 95 % avoidance rates respectively (Table S6). Griffon vultures spent on average 23 % and 22 % of their time (during the breeding and non-breeding season respectively) flying at the rotor risk heights, and 53 % of their overall time, flying within the 200 m buffer zone around the CWP (Table S4, Table S5). We predicted an overall annual Griffon vulture collision mortality for the CWP of 15 deaths under 98 % avoidance rate (Fig. 3; Table S4), varying from four up to 38 deaths under 99.5 % to 95 % avoidance rates respectively (Table S7).

In the future development scenarios, if all turbines operated simultaneously (CWP and FWP totalling 1284 turbines), the predicted cumulative annual collision mortality would account for 30 % of the

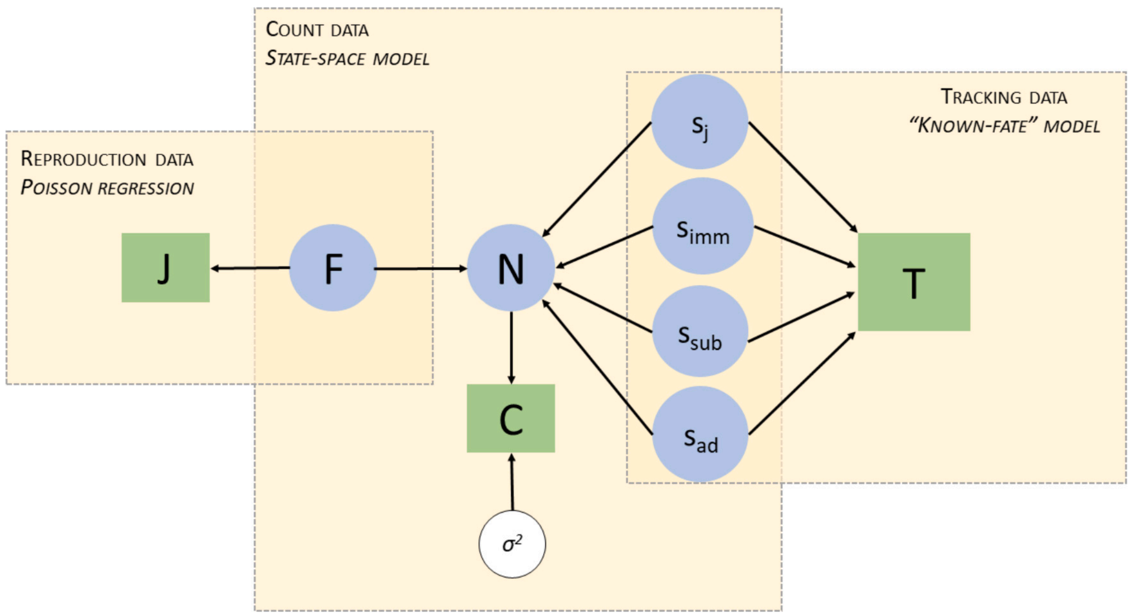

Fig. 1. A graphical depiction of the Integrated population models for the Cinereous and Griffon vultures. Demographic parameters are represented by blue circles, observation parameters with white circles, and data with green rectangles. Dependences between nodes are depicted using arrows. Submodels are depicted by large yellowish rectangles with dashed outlines. Data nodes included productivity (J), counts of occupied territories (C), and tracking data (T). Demographic parameters included fecundity (F), total number of individuals (N) and survival (s) for each stage (juveniles, immatures, subadults and adults). Parameters include observation error for count data (σ2). Figure adapted from Schaub and Abadi (2011).

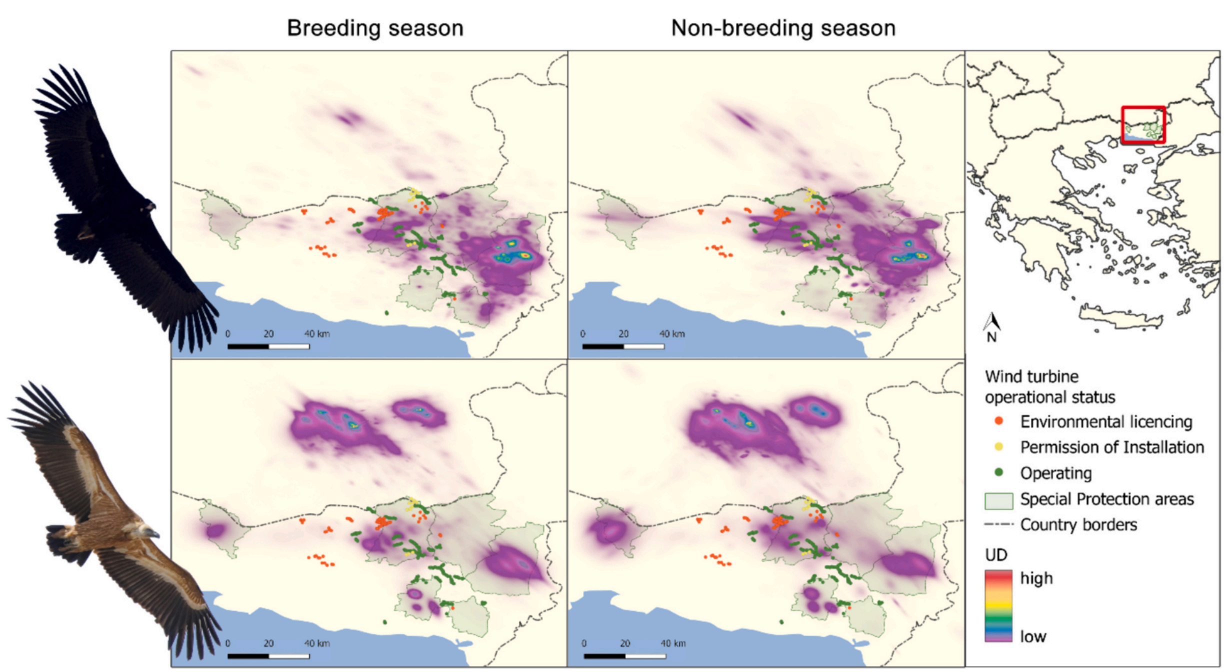

Fig. 2. Utilization Distribution (UD) of the Cinereous vulture (upper panels) and the Griffon vulture (lower panels) for the breeding and non-breeding season, calculated using Kernel density estimator on GPS telemetry data. The current wind power plants (CWP) categorised by operational status is shown along with their location in relation to Natura 2000 network (SPA: Special Protected Areas) in Greece.

current Cinereous vulture population and 7 % of the current Griffon vulture population (36 and 27 deaths respectively under 98 % avoidance rate; Fig. 3), under the most favourable (average-use) scenario. Under the least favourable scenario, i.e. use of FWP area according to the maximum observed use from CWP, the cumulative collision mortality doubled for the Cinereous vulture and raised by 1.6 times for the Griffon vulture (Fig. 3) compared to the most favourable scenario.

3.2. *Population trend and demographic parameters*

The Cinereous vulture population increased by 3.5 % per year between 1993 and 2022 corresponding to a mean replacement rate of 1.03 (95 % CI 0.96–1.12). Similarly, the Griffon vulture population increased by 18.4 % per year between 1989 and 2022 corresponding to a mean replacement rate of 1.07 (95 % CI 0.98–1.18). The dynamics of the Griffon vulture population seem to be driven mainly by adult survival as

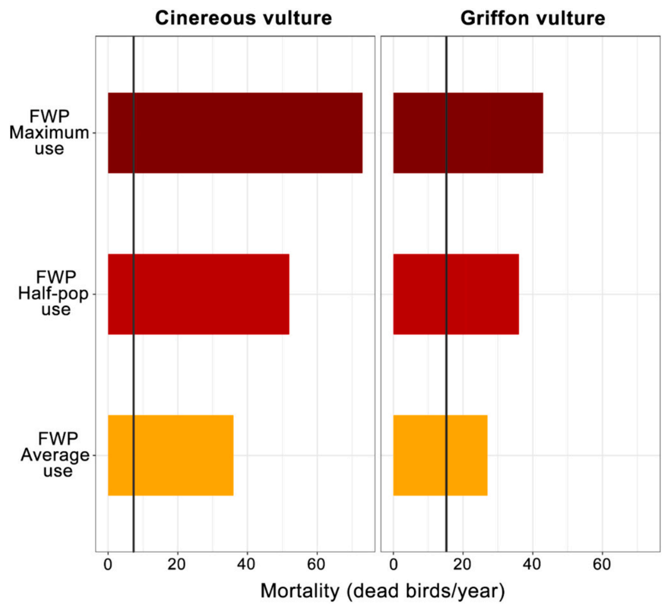

Fig. 3. Impact of future wind power plants (FWP) on the Cinereous and Griffon vulture populations, for three scenarios of space use (detailed description in section 2.3). Vertical line shows the mortality induced by the current wind power plants (CWP) for each species.

this was the main parameter that was found to be positively associated with replacement rate (Fig. S2). The integrated population model was able to replicate the trends and estimated most demographic parameters with reasonable precision (Table 2; Fig. S3; Fig. S4). Productivity was lower for the Cinereous vulture (mean ρ = 0.52, 95 % CI 0.40–0.67) compared to the Griffon vulture (mean ρ = 0.65, 95 % CI 0.56–0.74). Mean annual survival estimates were higher for adults for both species (>0.99), whereas mean annual survival of other age classes was overall lower for the Cinereous vultures, albeit with very wide credible intervals (Table 2). When both species' populations were projected 20 years in the future, the Cinereous vulture showed a slight decline of 12 % (from 32 pairs in 2022 to 28 pairs in 2042), whereas the Griffon vulture showed a continuous increase of 137 % (from 120 pairs in 2022 to 284 pairs in 2042). Without any increase in productivity and assuming stable survival rates, the populations' extinction probability was estimated at <0.3 % for both species in the next 20 years.

Table 2
Demographic parameter estimates of the Cinereous and Griffon vulture populations estimated with an integrated population model.

Parameter	Cinereous vulture			Griffon vulture		
	Mean	Lower 95 % credible limit	Upper 95 % credible limit	Mean	Lower 95 % credible limit	Upper 95 % credible limit
Productivity (ρ)	0.522	0.403	0.667	0.649	0.562	0.742
Annual survival juveniles (s_j)	0.791	0.238	0.991	0.879	0.492	0.995
Annual survival immatures (s_{imm})	0.831	0.307	0.994	0.912	0.624	0.997
Annual survival subadults (s_{sub})	0.817	0.285	0.995	0.915	0.632	0.997
Annual survival adults (s_a)	0.995	0.985	0.999	0.994	0.981	0.999

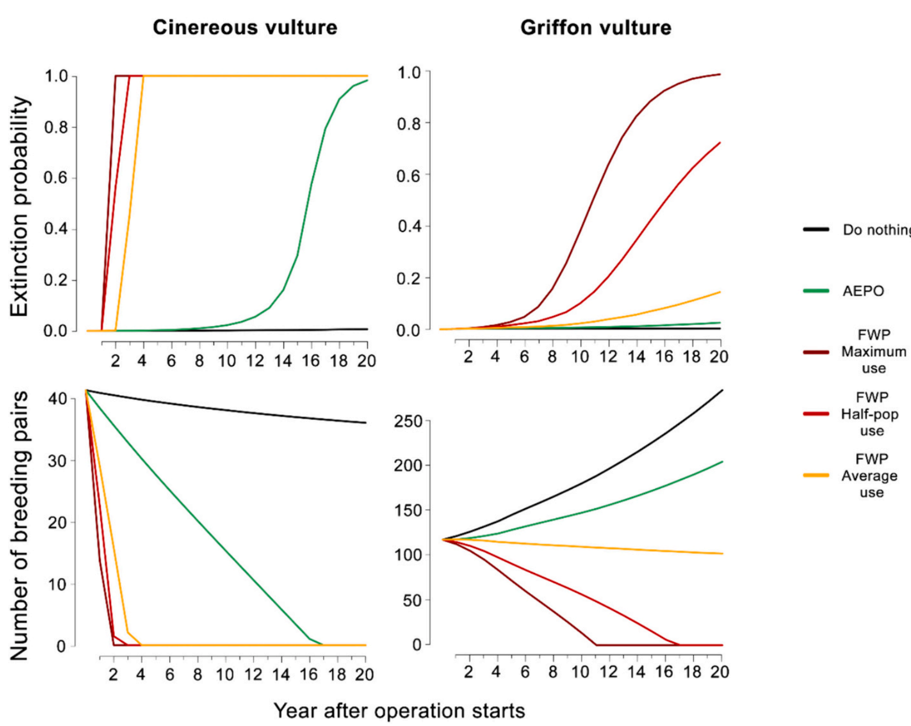

Fig. 4. Cumulative extinction probability and predicted population size of the Cinereous and Griffon vulture populations (mean values) when predicted 20 years into the future, for a baseline (do nothing) scenario, and four scenarios of wind energy development (detailed description in section 2.3).

3.3. Population-level consequences of future wind power plant establishment

If an additional 85 wind turbines (AEPO scenario, Table 1) start operating in the area, the Cinereous vulture population was predicted to go extinct within the next 18 years (Fig. 4). The Griffon vulture population was estimated to still increase, but with a lower rate, showing an increase of 67.5 %, from 120 pairs (CI 113–121) in 2022 to 201 pairs (CI 2–450) in 2042 (Fig. 4). The population extinction risk increased to 2.5 %. Our future scenarios assessing the impact of the operation of 934 additional wind turbines, located in the region would negatively impact the populations of both species, limiting population growth and resulting to extinction, depending on use intensity. For the Cinereous vulture, extinction is certain within five years of operation according to the low use scenario (FWP-average use scenario), within three years if half of the population uses the area (FWP-half-pop use scenario), and within two years of operation according to the high use scenario (FWP-maximum use scenario). For the Griffon vulture, a 13.6 % probability of extinction was estimated to be reached within 20 years from the start of operation, according to the low use scenario, leading to a slowly declining population at a rate of 0.6 % per year. When the area was assumed to be used by half of the population, extinction risk reached a probability of 71.4 % within 20 years, whereas the population was certain to go extinct after 20 years according to the high use scenario (Fig. 4).

4. Discussion

Our results are a quantitative warning on how additional mortality can impact the population viability of long-lived and protected species. We predicted high collision mortality rates for both vulture species, particularly for Cinereous vultures. Even under optimistic scenarios of future wind power plant development (high avoidance rates, average use of wind power plants), annual deaths could reach up to 30 % of the current Cinereous vulture population. Without further wind power plant expansion, both vulture populations were predicted to be stable or to increase over the next 20 years. However, planned wind energy development scenarios pose a serious threat: the operation of 85 additional turbines (already having acquired environmental licensing) would likely drive the Cinereous vulture to local extinction within 18 years, resulting in the devastating loss of this iconic species and the region's biodiversity. Griffon vultures would still experience population growth, but at a much-reduced rate. Scenarios with a larger number of turbines planned for the region would have a devastating impact on both populations, leading to extinction even under our most optimistic usage scenarios especially for the Cinereous vulture. Our results not only highlight the susceptibility of long-lived, K-selected species to any excess mortality, but also how it can severely impact even species with increasing populations (Sæther and Bakke, 2000; Duriez et al., 2023). However, it is essential to consider the potential influence of future immigration on population dynamics. Currently, immigration into the Cinereous Vulture population in Dadia National Park is minimal due to geographic isolation and limited connectivity with other populations. However, it is crucial to acknowledge that this situation may change. As Cinereous vulture populations in Bulgaria continue to grow the likelihood of immigration into our study area could increase in the mid-term future (10–15 years). Enhanced immigration could thus partially offset the excess mortality caused by wind turbine collisions, thereby slowing the negative impacts on population dynamics. On the other hand, increased immigration could also result in the local population acting as a demographic sink, having serious consequences for the overall species if the source populations are not robust enough to sustain continued losses. Future studies based on our model could be repeated to quantify the impact of restocking on the species population.

It is widely acknowledged that collisions with wind turbines can have enduring demographic impact on long-lived species including bats, seabirds and raptors (Carrete et al., 2009; Frick et al., 2017; Martín et al., 2018; Lane et al., 2020). We expect that an abundant and increasing population will display greater resilience to increased mortality than a small declining one and our results confirm that hypothesis. The retrospective analysis showed a much slower growth rate (3.5 % annually) for Cinereous vulture compared to Griffon vultures (18.4 % annually). This difference can be attributed only in part to the lower reproductive rates observed in Cinereous vultures when compared to other populations across its distribution (Andevski et al., 2017). However, survival rates for juveniles were found to be low, while immature and adult survival was similar compared to other populations (Rousteau et al., 2022). The Cinereous vulture already faces major threats in its breeding grounds, such as the illegal use of poison baits (Hernández and Margalida, 2008; Skartsi et al., 2008; Skartsi et al., 2014; Sanz-Aguilar et al., 2015) and electrocution, that may act in synergy with wind turbine induced mortality (Vasilakis et al., 2016) thus restricting population increase. The same threats also occur for the Griffon vulture in the region. The trans-border population's productivity was found to be lower than the one reported solely for the Bulgarian population for the period 1987–2011 (Demerdzhiev et al., 2014), but still higher than that of the newly-established colonies of reintroduced individuals in Bulgaria (Kmetova-Biro et al., 2021) or insular populations in Greece (Xirouchakis, 2010) and comparable to the species productivity in Spain and Portugal (Monteiro et al., 2018). Survival was high, especially for adults, compared to other populations across the species range (Arrondo et al., 2020; Monti et al., 2023). Although the long-term stability of the Griffon vulture populations could be strongly affected by juvenile survival (Van Beest et al., 2008), the dynamics of this population seem to be driven mainly by adult survival. Therefore, from a demographic point of view, a priority task for the successful conservation of both species is the reduction of non-natural mortality rather than trying to increase their productivity. This strategy has also been proposed for other long-lived species (Ortega et al., 2009). Several studies across regions have reported significant impacts of wind power plants on soaring birds. In Spain, wind farms contribute to substantial mortality rates in raptors, particularly the endangered Spanish imperial eagle (Carrete et al., 2009). In the U.S., similar impacts have been documented on Golden eagles (Katzner et al., 2017). Furthermore, 26 % of the European population of the Lesser kestrel has at least one wind turbine within the foraging areas around colony sites (Assandri et al., 2024), This study further builds on the observed consistent pattern of vulnerability of raptor species that rely on extended territories and specific flight behaviors. Thus, the mitigation strategies we propose for the Cinereous and Griffon vultures could be applicable to other species in different geographic contexts.

Current and predicted collision mortality rates (measured as collisions per turbine per year) for vultures in our study area are within the range reported for other soaring birds (Drewitt and Langston, 2006; De Lucas et al., 2008). Although the rates of collision mortality often appear small, the variance is very large and thus could mask the much higher collision rates for individual turbines or groups of turbines (Drewitt and Langston, 2006). The impact of this excess mortality is better reflected in the number of birds killed per year, especially when reported as the percentage of the population affected. The Lesser kestrel population in France experienced a mortality rate of approximately 3 % per year due to wind turbine collisions (Duriez et al., 2023), Red Kite fatalities at wind turbines operating in Germany represent the 3.1 % of the estimated post-breeding population of the species (Bellebaum et al., 2013), whereas in the Netherlands, although 2.5 % of the Montagu's Harrier population is currently affected per year, this percentage could reach 12 % under future wind power plant development (Schaub et al., 2020). We estimated one of the highest impacts recorded, as direct mortality from wind turbines is already affecting 6.7 % and 3.9 % of the Cinereous and Griffon vulture populations respectively; under the most favourable scenario of future wind power plant development, 30 % and 7 % of the Cinereous and Griffon vulture populations could perish annually and therefore, cumulative annual collision mortalities for current and future wind power plants are far too high for the populations of the study species to be sustainable. It should be further noted that this high variance in collisions per wind turbine per year could reflect the local conditions shaping the movements of birds. In the Bulgarian side of the Eastern Rhodopes currently there are no operating wind power plants but there is a growing interest, and such projects might appear in near future which will have cumulative impact on the populations of both species. It should be noted that in addition to the direct mortality from wind turbine collisions, another potential threat to vulture populations in our study area might arise from ecological traps caused by the presence of livestock carrion in the environment. Integrating carcass removal into vulture conservation strategies can help reduce the attractiveness of high-risk areas, thereby lowering the collision risk (Martin et al., 2012; Bennun et al., 2021).The general movement patterns of both vulture species overlapped in certain areas, validating the existence of previously depicted movement corridors (Vasilakis et al., 2016), thus indicating that wind power exclusion zone proposals of previous studies in the area could effectively contribute to the survival of the species (WWF Greece, 2013; Vasilakis et al., 2017). Such approaches are mentioned as best practices in the EU Guidance Document on wind energy developments and nature legislation and must be considered in the forthcoming revision of the Special Framework for Spatial Planning for the Development of Renewable Energy Sources and Land Management.

This research introduces a number of advanced approaches to assess the impact of wind energy development on vulture populations. Since the presence of wind power plants can have substantial effects on species with large home-ranges (Carrete et al., 2009), it is imperative to adopt methodological approaches that consider space use by the species throughout their annual cycle (Masden and Cook, 2016). Our comprehensive approach to collision rate estimations, aggregates seasonal mortality rates to present a year-round assessment of collision risks, further enhancing the understanding of the cumulative impacts on vulture populations. Moreover, our emphasis on cumulative impacts, extending beyond individual protected areas, takes into account the entire foraging range of vulture species providing a more realistic assessment of long-term consequences. Despite the fact that there is some uncertainty surrounding model projections, still, by employing Integrated Population Models, we provide a comprehensive prediction of population dynamics, taking into account the simultaneous effects of multiple sources of uncertainty that simpler deterministic models often overlook (Oppel et al., 2014). Additionally, we capture real-world variability and environmental uncertainty faced by this small population, in contrast to previous approaches (Dimitriou et al., 2021). However, one of the most significant sources of uncertainty lies in the avoidance rate parameter used in the CRM. While we based our IPM scenarios on a 98 % avoidance rate— as the most realistic for our study species based on previous studies (Vasilakis et al., 2016)—this value is difficult to estimate with precision. Small deviations from this assumption can lead to substantial variation in predicted mortality as we show in the testing of a range of avoidance rates from 95 % to 99.9 % in out CRM. It is important to further note that although our collision models did not explicitly account for possible collision deterrent measures, these are actually incorporated in the CRM, and subsequently the IPMs, by accounting for the time that vultures spend in the 200 m buffer zone around the turbines, where deterrents are used. Still, such acoustic deterrents although showing some initial promise in reducing bird collisions, their effectiveness may be limited due to habituation (Marques et al., 2014), and they require further validation through rigorous on-ground testing using a BACI design to confirm their long-term effectiveness. Furthermore, mitigation measures such as selective stopping, and blade painting, have shown significant potential in reducing collision risks and need to be further validated through field studies (May et al., 2020; Ferrer et al., 2022).

Our study provides practical implications for EIAs of wind power development because our dynamic modelling approach highlights the demographic impact of excess mortality on wild populations. Such models are becoming increasingly important since the continuous increase of wind power plants leads to growing concerns about their cumulative impacts on avian populations (Masden et al., 2010; Masden et al., 2012). Therefore, before the endorsement of any wind power plant project, EIAs should -and must- run population dynamics analyses to assess long-term demographic impacts on the population scale and also addressing potential cumulative impacts of all wind power plants in the vicinity of the assessed project, rather than merely recording short-term mortality at the individual level, as is often promoted by the wind power industry (Enríquez-de-Salamanca, 2018). Although our IPMs were parameterised using accurate long-term datasets of population counts, breeding success and GPS/GSM tracking, it is understandable that such datasets are not always available, thus limiting such approaches. Despite that, recent approaches do exist that require minimal input data about the focal population and can simulate and assess the impact of various mortality sources (Chambert et al., 2023). Such tools, designed for professionals and decision makers involved in drafting EIAs), aim to enhance their quality by quantifying the expected impacts of a project at the population level making a step towards reducing decision biases (Williams and Dupuy, 2017).

5. Conclusion

Our study shows how wind energy development can cause significant negative cumulative impacts on vulture populations. The expected collision mortality rates predicted the extinction of the endangered Cinereous Vulture in all tested scenarios and of both target vulture species under the high use scenarios. For Griffon vultures, a severe decline is expected even under our most optimistic scenarios. This decline could have substantial effects on the important ecosystem services provided by this species, as scavengers play a crucial role in the ecosystem by consuming carrion, which helps to control disease and recycle nutrients, contributing to overall ecosystem health (Buechley and Şekercioğlu, 2016). In India, the decline of vulture populations due to poisoning has led to an increase in feral dog and rat populations, which has been linked to rising cases of rabies in humans (Markandya et al., 2008). A similar decline in scavenging efficiency in our study area could result in negative impacts on public health and ecosystem stability. Thus, the conservation of these species extends beyond biodiversity and is crucial for maintaining ecosystem services that benefit human populations. To ensure the long-term persistence of these vulture populations, a comprehensive conservation approach is crucial, integrating both species conservation efforts and strategic planning for wind energy development. Rigorous pre- and post-construction monitoring using standardized methods is essential to assess actual impacts and inform future assessments. Results of these monitoring efforts should be publicly available so that they can become easily accessible to fellow assessors that may be working in adjacent sites, thus facilitating data availability for quantifying the expected impacts of a project at the population level. Furthermore, cumulative impact assessments must encompass the entire vulture foraging range across their life cycle, and not only be restricted within individual protected areas and further consider vulture social behavior to balance their conservation with anthropogenic activities (van Overveld et al., 2020). Balancing renewable energy development with the preservation of biodiversity requires careful planning, strategic decision-making, and ongoing monitoring to adapt conservation strategies as needed. Wind turbine related mortality can be lowered by removing turbines associated with high collision risk and by planning the installation of future wind turbines outside of areas that are critical for vultures. At the buffer zone of those areas, where the birds are still present, but less frequently, strict mitigation measures should be adopted, following international guidelines and relevant literature (Bennun et al., 2021; European Commission, 2020). Overall, this study emphasizes the need to integrate biodiversity considerations into renewable energy planning. Approaches towards the increase of renewable energy production, to meet relevant targets, while keeping conservation cost to a minimum, have already been suggested (Vasilakis et al., 2017; Kati et al., 2021) and should be taken into consideration during spatial planning for the development of renewable energy sources.

Ethics statement

All procedures regarding animal manipulation and tagging complied with permits and regulations issued by the Hellenic Ministry of Environment and Energy and the Bulgarian Ministry of Environment and Water.

Funding

This research was developed in the frame of the study for cumulative effects of wind power in Thrace implemented by the Society for the Protection of Biodiversity of Thrace under the Project "SafeFlyways-reducing energy infrastructure related bird mortality in the Mediterranean" funded by MAVA Foundation and BirdLife International. The long-term monitoring of the Cinereous vulture in Greece was implemented by WWF Greece. The monitoring of the Cinereous vulture breeding colony during 2014–2021 and the monitoring of vultures at the feeding site during 2016–2021 as well as tracking data collection for estimating collision risk, were funded under Operational Programmes "Environment and Sustainable Development" (ESD) 2007–2013 and "Transport Infrastructure, Environment and Sustainable Development (TIESD)" 2014–2020, co-funded by the European Union. The long-term monitoring of the Griffon vulture colonies in Bulgaria and Greece was implemented by the Bulgarian Society for the Protection of Birds/BirdLife Bulgaria and WWF Greece. The fieldwork, tracking data and data collection for period 2016–2021 were conducted in the framework of the LIFE Project "Conservation of Black and Griffon vultures in the cross-border Rhodope Mountains" (LIFE14 NAT/NL/000901) funded by the European Commission and supported by Rewilding Europe and Segre Foundation.

CRediT authorship contribution statement

Anastasios Bounas: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Visualization, Writing – original draft. **Dimitrios Vasilakis:** Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Writing – review & editing. **Elzbieta Kret:** Conceptualization, Investigation, Data curation, Writing – review & editing. **Sylvia Zakkak:** Investigation, Data curation, Writing – review & editing. **Yorgos Chatzinikolaou:** Methodology, Investigation, Writing – review & editing. **Eleftherios Kapsalis:** Investigation, Data curation, Writing – review & editing. **Volen Arkumarev:** Investigation, Data curation, Writing – review & editing. **Dobromir Dobrev:** Investigation, Data curation, Writing – review & editing. **Anton Stamenov:** Investigation, Data curation, Writing – review & editing. **Stoycho Stoychev:** Investigation, Data curation, Writing – review & editing. **Theodora Skartsi:** Conceptualization, Investigation, Data curation, Writing – review & editing, Project administration, Funding acquisition. **Lavrentis Sidiropoulos:** Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Writing – review & editing. **John M. Halley:** Conceptualization, Methodology, Validation, Formal analysis, Data curation, Writing – review & editing, Project administration.

Declaration of competing interest

The authors have no conflicts of interest to declare.

Data availability

Tracking data are available through Movebank (https://www.movebank.org/) via the Movebank IDs: 126437978, 219704159, 219687539. IPM code available in Zenodo (https://doi.org/10.5281/zenodo.11408329).

Acknowledgements

We would to like to thank P. Babakas and K. Pistolas from the Evros Regional Unit for their crucial contribution and collaboration for the implementation of the joint annual monitoring of the Cinereous vulture breeding population in Dadia-Lefkimi-Soufli Forest National Park. We thank all the employees of the Management Unit of Evros Delta and Dadia National Parks for their support in vulture monitoring and administrative issues, and especially I. Tziambazis and A. Chalivelentzios, who have been undertaking the fieldwork for the monitoring of the Cinereous vulture breeding population since 2015. We would like to express our gratitude to all volunteers and experts of BSPB who supported the monitoring and tagging of Griffon Vultures in Bulgaria and especially: Atanas Delchev, Nikolay Terziev, Vera Alekseeva, Vladimir Dobrev, Emil Yordanov, Ivaylo Angelov, Sylvia Dyulgerova, Desislava Kostadinova, Hristo Hristov, Mario Keller, Dimitar Nedelchev, Vanya Angelova.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eiar.2024.107669.

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