Large scale risk-assessment of wind-farms on population viability of a globally endangered long-lived raptor

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A B S T R A C T

Wind-farms receive public and governmental support as an alternative energy source mitigating air pollution. However, they can have adverse effects on wildlife, particularly through collision with turbines. Research on wind-farm effects has focused on estimating mortality rates, behavioural changes or interspecific differences in vulnerability. Studies dealing with their effects on endangered or rare species populations are notably scarce. We tested the hypothesis that wind-farms increase extinction probability of long-lived species through increments in mortality rates. For this purpose, we evaluate potential consequences of wind-farms on the population dynamics of a globally endangered long-lived raptor in an area where the species maintains its greatest stronghold and wind-farms are rapidly increasing. Nearly one-third of all breeding territories of our model species are in wind-farm risk zones. Our intensive survey shows that wind-farms decrease survival rates of this species differently depending on individual breeding status. Consistent with population monitoring, population projections showed that all subpopulations and the meta-population are decreasing. However, population sizes and, therefore, time to extinction significantly decreased when wind-farm mortality was included in models. Our results represent a qualitative warning exercise showing how very low reductions in survival of territorial and non-territorial birds associated with wind-farms can strongly impact population viability of long-lived species. This highlights the need for examining long-term impacts of wind-farms rather than focusing on short-term mortality, as is often promoted by power companies and some wildlife agencies. Unlike other non-natural causes of mortality difficult to eradicate or control, wind-farm fatalities can be lowered by powering down or removing risky turbines and/or farms, and by placing them outside areas critical for endangered birds.

1. Introduction

Greenhouse gas emission is the primary cause of anthropogenically driven global climate change (Huntley et al., 2006), and wind-farms represent a relatively new source of energy mitigating air pollution associated with fossil fuel technologies (Nelson and Curry, 1995). Thus, they have received strong public and governmental support as an alternative energy source (Leddy et al., 1999). However, wind-farms can have adverse effects on wildlife, particularly through bird and bat collision with rotating turbine rotor blades (e.g., Langston and Pullan, 2003; Baerwald et al., 2008).

Population viability analyses are increasingly used to provide an ecological basis for decision-making and, therefore, to guide management actions for rare or endangered species (e.g., Lindenmayer and Possingham, 1996; Carrete et al., 2005; Oro et al., 2008). Debate on the effects of human activities on wildlife such as those related to wind-farm developments are particularly in need of these types of risk and impact assessments. However, efforts toward this end have been largely directed toward estimating annual mortality rates of different species or taxonomic groups (Smallwood and Thelander, 2008) and toward assessing behavioural changes (Larsen and Guillemette, 2007) or interspecific differences in vulnerability to wind-farms (Garthe and Hüppop, 2004). Studies dealing with long-term population effects of wind-farm mortality are notably scarce, even when current modelling procedures might allow us to obtain reliable forecasts of the impact of these human developments on population dynamics still when only poor data sets are available. In this sense, Population Viability Analysis (PVA) are highly useful to assessing trade-offs in data-poor cases while contributing to precautionary actions and management decisions (Thompson et al., 2001; Tuck et al., 2001; Cooney, 2004; Curtis and Vincent, 2008).

Spain is the world's third largest wind-power producer after the United States and Germany, with more than 640 wind-farms consisting of ca. 14,000 turbines, which produce 15,154MW of

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generating capacity (http://www.aeeolica.es). At the same time, this country is a region vastly important to wildlife, with populations strongholds of many threatened European avian species (Birdlife International, 2000). Thus, as occurred some years ago with the expansion of electric power lines (Ferrer and Janss, 1999), the effects of wind-farms on species of conservation concern, such as many raptors, should be carefully monitored and our "progress" reconciled with biodiversity conservation (Tellería, 2009a,b). Indeed, some of the highest levels of mortality at wind-farms have been for this group, and different studies suggested that both migrating birds and those resting and foraging locally are affected (Barrios and Rodríguez, 2004; Madders and Whitfield, 2006).

Our study hypothesis is that wind-farms increases extinction probability of long-lived species through increments in mortality rates. For this purpose, we evaluate consequences of wind-farm development on the population dynamics of an endangered long-lived raptor, the Egyptian vulture *Neophron percnopterus*. The populations of this cliff-nesting bird have steadily declined over large parts of its European, African and Asian range during the 20th century. In peninsular Spain, where the bulk of its breeding population is located (ca. 80%; Donázar, 2004), 25% of its breeding territories recently became extinct (Carrete et al., 2007) and the species is thus regarded as 'endangered', both in Spain (Donázar, 2004) and globally (Birdlife International, 2008; IUCN, 2008). Abandoned territories of Egyptian vultures have been found to be aggregated in extinction 'hotspots', mainly related to food availability, human pressure (mainly illegal poisoning and ingestion of antibiotics from livestock), and isolation from other conspecific territories (Carrete et al., 2007; Blanco et al., 2007). Now, another threat can be included in this list, with alarming numbers of Egyptian vultures found dead in the vicinity of wind-farms (e.g., in 2008, eight individuals found dead in wind-farms of Southern and Northern Spain; Birdlife International, 2008). Although it may be extremely difficult to exactly predict future population impacts of wind-farms on this vulture, even a crude picture of the extent to which these human facilities may represent a potential threat is of great interest to managers and policy makers throughout the worldwide distribution of the species (Birdlife International, 2008). Moreover, results can then be extended worldwide to the management of other endangered, long-lived species for which less demographic information is available, such as golden eagles *Aquila chrysaetos*, Bonelli's eagles *Hieraaetus fasciatus*, black storks *Ciconia nigra*, or red kites *Milvus milvus*, and which are also experiencing increased mortality rates at wind-farms in other countries of Europe or in the US (Hunt et al., 1999; Barrios and Rodríguez, 2004; Kuvlesky et al., 2007; de Lucas et al., 2008).

2. Methods

2.1. Study species

The Egyptian Vulture is a medium-sized, cliff-nesting, trans-Saharan migrant raptor that defends long-term established territories during the breeding season. Most territories hold a single nest (rarely 2–3 nests situated in the same or adjacent cliffs) that is occupied year after year over long periods of time. The long-term monitoring of marked birds shows that territories are reoccupied every year in early March by their previous owners or, when one dies, by a replacement bird (J.A. Donázar, J.R. Benítez, J.A. Sánchez-Zapata, J.L. Tella, J.M. Grande, unpublished data; see below). Recruitment typically takes place at 6 years of age, and during the non-breeding stage, at least while in Europe, Egyptian vultures visit predictable food sources and gather in communal roosts (Carrete et al., 2007), moving all over their natal breeding areas (J.A. Donázar, M. Carrete, A. Cortés, J.M. Grande, unpublished data). The species shows differential maturity and a variety of plumages that allow us to confidently assess their age before adulthood (5 years).

Although information on dispersal rates of the species remains scarce, data on individually marked birds suggest that Egyptian vultures are largely philopatric and faithful to their breeding territories (Grande, 2006). Indeed, natal dispersal distances are relatively short (36.39±42.48 km; range = 0–150.52 km; $n = 22$) and breeding dispersal can be considered as near null (only in 7.5% of 203 breeding attempts of individually marked birds recorded across peninsular Spain did one of the breeders move to a neighbouring breeding territory which were always located at <5 km; J.A. Donázar, J.M. Grande, J.L. Tella, unpublished data). These and other sources of evidence suggested that the Spanish population could actually be behaving as a meta-population divided into at least three main subpopulations (Fig. 1a).

2.2. Data collection

We used information from an intensively surveyed subpopulation to estimate minimum mortality rates of territorial and non-territorial birds associated with wind-farms. Then, we extended these results to the entire Spanish distribution of the species to model potential population outcomes on a large spatial scale (see Section 3).

The Strait of Gibraltar (Fig. 1b) is included among the four areas in Spain with the greatest potential for producing wind-energy. There, wind-farms have been monitored since 1993 by power companies and local governments, such that a record of the number, date, location and causes of death (established by veterinarians of the Wildlife Forensic Laboratory of the Junta de Andalucía) of Egyptian vultures found dead is available (Diputación de Cádiz and Junta de Andalucía). At the same time, all geographic positions of turbines ($n = 675$) were obtained from current satellite images of the study area so that distance from bird territories to point of death can be accurately calculated.

From 2000 to 2008, we intensively surveyed territories of Egyptian vultures in this area and its surroundings, all of them included within the southern core of the species in peninsular Spain (Fig. 1). Breeding territories were intensively monitored (range: 3–7 visits/breeding period) to estimate productivity (number of fledglings) as well as to confirm their occupation by breeding birds. Otherwise, adult absences were assertively detected almost weekly. At the same time, wind-farm monitoring for bird carcasses was intensified.

Searches for bird fatalities around each turbine were carried out at standardized intervals (once a week) in 27 out of 29 wind-farms with surveillance located in the study area (12,000 km^2). However, we intensively searched for birds when an adult bird was not present in its territory. Thus, we were quite confident in our assessment of breeding bird mortality associated with wind-farms. Mortality of non-breeding birds was less confidently obtained since individuals were found but not actively searched for. Following de Lucas et al. (2008), no corrections for corpses that were overlooked or removed by scavengers were applied, so our data may underestimate the mortality rate of Egyptian vultures associated with wind-farms. However, these authors stated that although decomposition occurred over time, remains are present in the study area for months to years, a period much longer than any inter-search interval.

The large-scale distribution of Egyptian vultures was obtained by using the results of the 2nd Spanish Survey of the species performed by more than 600 experienced local ornithologists during 2000 (for details on survey methods, please refer to Carrete et al., 2007). Although this information may be slightly dated (1279 occupied and 433 breeding extinct territories in 2000; Fig. 1b), it

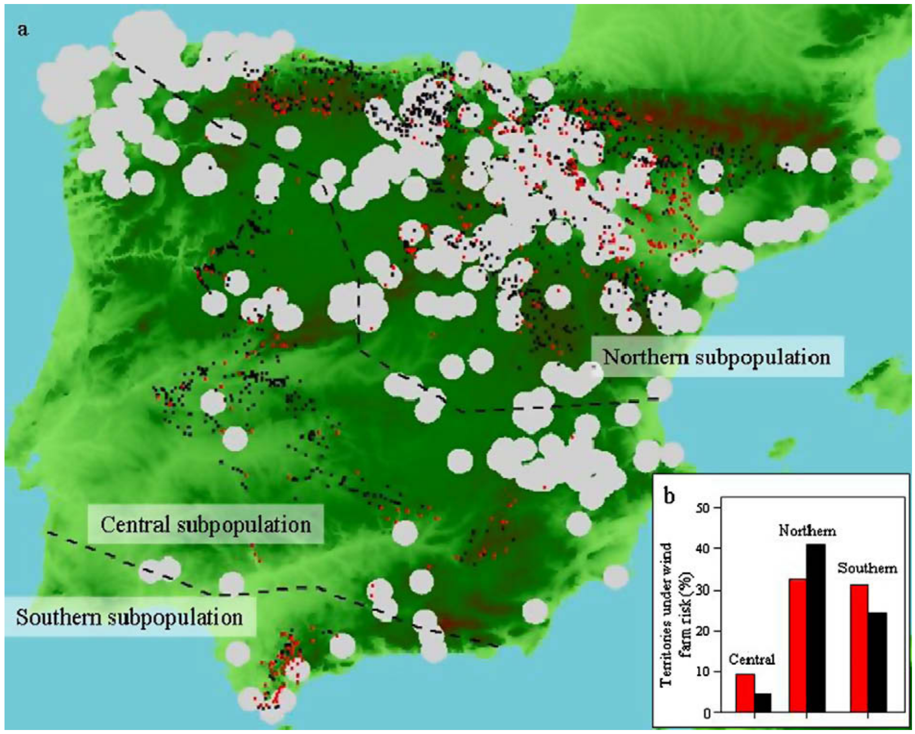

Fig. 1. (a) Distribution of occupied (black dots) and vacant (red dots) territories of Egyptian vultures and high-risk areas associated with wind-farms (grey) in peninsular Spain. Dashed lines separate the three main subpopulations constituting our meta-population. (b) Percentage of occupied (black bars) and vacant (red bars) territories of Egyptian vultures situated within the wind-farm risk zones (<15 km from wind-farms). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

is useful for our purposes since it corresponds to the entire breeding range of the species. Indeed, in the few areas in which the species is intensively surveyed, new territories always consist of recolonization of extinct ones (Carrete et al., 2007).

Large-scale information on wind-farms was obtained online from the Spanish Wind Energy Association (http://www.aeeolica.es), using both their estimated location as plotted on their webpage or their presence in a municipal district (using its centroid as an approximate location). All points were buffered using a radius of 15 km (i.e. the maximum distance at which a wind-farm killed a territorial Egyptian vulture; J.A. Donázar, J.R. Benítez, J.A. Sánchez-Zapata, unpublished data) to achieve risk zones for the species (Fig. 1a, Table 1). To simplify analysis, no distinction associated with the size of each wind-farm (i.e., number of turbines or surface area affected) was made.

2.3. Demographic parameters

A recent study using individually marked birds shows that survival probabilities for Egyptian vultures are age-dependent (Grande et al., 2008). Survival increased with age until birds acquired their adult plumage and searched for a breeding territory, at which point it decreased. At older ages (>6 years), survival was higher for both non-breeding and breeding adults (Table 2). Other aspects such as natal and breeding territory quality (measured through their normalized difference vegetation index, *NDVI*, and their mean long-term productivity, respectively; Grande et al., 2008) also affect survival rates. Thus, baseline population projections were performed using age-specific survival rates obtained by replacing information from each subpopulation on survival models available for the species (Grande et al., 2008). Reproductive rates (i.e., percentage of females breeding successfully, and percent of broods with one or two offspring) were estimated from 2,470 breeding events recorded across subpopulations during 1983–2008 (García-Ripollés and López-López, 2006; G. Blanco, A. Margalida, I. Zuberogoitia, J.M. Grande, O. Ceballos, A. Cortés-Avizanda, Quique, J.A. Donázar, J.R. Benítez, J.A. Sánchez-Zapata, unpublished data).

In the intensively surveyed subpopulation (the southern one; Fig. 1), mortality of territorial birds associated with wind-farms was obtained by considering the number of breeders found dead in these developments out of the total breeding population under risk (i.e., number of breeders occupying territories within the buffer risk zone). Mortality of non-territorial birds was also calculated using information on non-breeding birds found dead in wind-farms out of the total non-breeding population size esti-

Table 1
Subpopulation sizes and potential wind-farm risk for the Egyptian vulture in peninsular Spain. In brackets, percentage of breeding territories.

	Subpopulation size		Wind-farm risk		
	No. of breeding territories	No. of non-breeding birds	Occupied territories under risk	Vacant territories under risk	Surface under risk (%)
Subpopulations					
North	922	1685	376 (41)	101 (32)	38.75
Centre	328	1050	15 (5)	6 (9)	18.46
South	29	58	7 (24)	17 (31)	16.78

Table 2
Parameters used to simulate meta-population trends considering (1) survival rates of territorial and non-territorial birds without wind-farm effects and constant across subpopulations (null model), and (2) survival rates of territorial and non-territorial birds affected by wind-farm mortality (wind-farm model). Standard deviations are in brackets.

	Dispersal rates			Reproductive rates		Survival rates			
						Null model		Wind-farm model	
Subpopulations	North	Centre (to)	South	% of females breeding successfully	% of broods with 1 offspring	Territorial birds	Non-territorial birds	Territorial birds	Non-territorial birds
							1–2 y: 0.73 (0.02)		1–2 years: 0.72 (0.02)
		High: 0.183	High: 0			0.894	3–4 y: 0.78 (0.01)	0.889	3–4 years: 0.77 (0.01)
North		Medium: 0.0001	Medium: 0	58.85	60.36				
		Low: 0	Low: 0			(0.02)	5 y: 0.60 (0.04)	(0.02)	5 y: 0.59 (0.04)
							>6 y: 0.75 (0.02)		>6 y: 0.74 (0.02)
	High: 0.31		High: 0.02				1–2 y: 0.73 (0.02)	0.891	1–2 y: 0.72 (0.02)
Centre (from)	Medium: 0		Medium: 0	63.57	71.19	0.891	3–4 y: 0.78 (0.01)		3–4 y: 0.77 (0.01)
	Low: 0		Low: 0			(0.02)	5 y: 0.60 (0.04)	(0.02)	5 y: 0.59 (0.04)
							>6 y: 0.75 (0.02)		>6 y: 0.74 (0.02)
	High: 0	High: 0.702					1–2 y: 0.73 (0.02)		1–2 y: 0.72 (0.02)
South	Medium: 0	Medium: 0		60	78.86	0.895	3–4 y: 0.78 (0.01)	0.892	3–4 y: 0.77 (0.01)
	Low: 0	Low: 0				(0.02)	5 y: 0.60 (0.04)	(0.02)	5 y: 0.60 (0.04)
							>6 y: 0.75 (0.02)		>6 y: 0.74 (0.02)

mated (see below). It is worth noting that individuals from other subpopulations may occupy this area during migration (from late February to early June, and from mid August to late September; Benítez et al., 2009). However, non-breeding birds were found dead outside of these periods so we were confident that this mortality can be assigned to the local non-breeding fraction. Extrapolation of this impact of wind-farms on Egyptian vulture mortality to the other two subpopulations was done by using as a reference their percentage of occupied breeding territories (for territorial bird mortality) or surface area (for non-territorial bird mortality) exposed to wind-farm risks (see before; Table 1).

Actual dispersal rates among subpopulations are unknown, so we performed our simulations by including three potential scenarios of dispersal rates, corresponding to minimum (excluding zero values), average, and maximum natal dispersal distances observed for the species in one of the two largest subpopulations (2.47, 36.39, and 150.52 km, respectively; Grande, 2006). Dispersal rates were obtained as the proportion of territories within each subpopulation that can "export" a bird to another subpopulation (Table 2). That is, the proportion of territories within an "exporting" subpopulation is near a potentially vacant territory of the "importing" subpopulation (i.e., at a distance $\leqslant$ to the dispersal distance considered). As these values included in population projections were arbitrarily obtained, we included this parameter (dispersal rate) as a random term in models comparing the effects of wind-farm mortality on population viability estimates (population size and probability of extinction; see below). Finally, total subpopulation sizes were calculated by summing breeding and non-breeding birds of the different age-classes as $2 * nBT + Prod * nBT + \Sigma(Prod * nBT * S_n)$ where nBT is the number of breeding territories of the subpopulation, $Prod$ is its mean productivity, and S_n is the survival of individuals at the different age-classes (see Fig. 2).

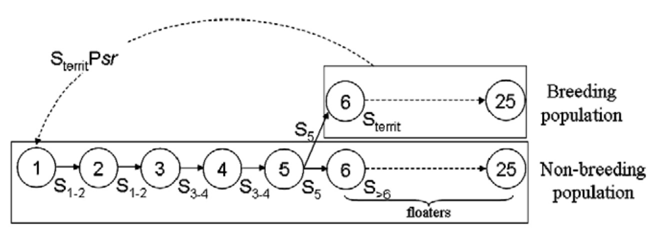

Fig. 2. Life cycle of the Egyptian vulture meta-population. Nodes represent the different age-classes considered in the models (from 1 to 25 years old), S_i are survival rates of each age-class i; P: productivity. The model is only for females.

2.4. Modelling procedures

We used the program VORTEX V9, an individual-based simulation modelling tool widely used for population viability analysis (Lacy et al., 2003), to build prospective stochastic age-structured population models (only for females) to simulate the effects of wind-farm mortality on the population dynamics of Egyptian vultures. For this purpose, we obtained a null model without wind-farm effects by using survival rates of territorial and non-territorial birds obtained for the species through Capture-Mark-Recapture models (Grande et al., 2008; Table 2). Models including wind-farm effects were made by including our estimations of mortality rates of territorial and non-territorial birds associated with these infrastructures. Reproductive rates were the same for all scenarios (Table 1). Simulations were run over 100 years for 100 different iterations. Previous studies carried out in the northern subpopulation suggest that density-dependent process are not present, maybe as a consequence of population decline (Grande, 2006); thus, we did not include this effect during modelling. We considered that a population became extinct when only one individual remained alive. Mean population size (i.e., number of individuals) and probability of extinction (i.e., proportion of iterations run before a population became extinct) after 100 simulated years were obtained as estimates of subpopulation and meta-population viability to make comparisons between our two scenarios (null or wind-farm model; fixed factor) through Generalized Linear Mixed Models (Mean population size: error distribution: normal, and link function: identity; Probability of extinction: error distribution: binomial, and link function: logit). As dispersal rates were our less certain parameter, we included it as a random term in models. Time since the beginning of the simulation (in years) was included in models as a covariate to control for non-independence of data.

3. Results

3.1. Wind-farm mortality rates

From 2004 to 2008, we found two territorial and three non-territorial (two young and one individually marked floater 6 years old) birds dead in wind-farms located in our intensively surveyed subpopulation, the southern one (Fig. 1). Nearest distance between breeding territories suffering from these casualties and wind-farms were 6.37 and 14.57 km, so we used 15 km as a guiding radius to obtain wind-farm risk zones (see below).

The minimum annual mortality rate of territorial birds associated with wind-farms was 0.015±0.03 (range: 0–0.071). In this way, vultures occupying territories located at fewer than 15 km away from wind-farms have an extra probability of 0.015 of mortality due to collision with a turbine. The minimum annual mortality rate of non-territorial birds associated with wind-farms was lower (0.008±0.016; range 0–0.046). Thus, our intensive survey of the southern subpopulation of Egyptian vultures suggests that wind-farms can decrease the survival rates of the species, and that these impacts are different depending on individual breeding status (breeder or non-breeder).

3.2. Distribution of breeding territories and wind-farms

Nearly one-third of all territories known to have been occupied by Egyptian vultures in peninsular Spain in the late 1980s (n = 1712) were included within our wind-farm risk zones, territories occupied in 2000 representing the largest part of this percentage when compared with extinct ones (Table 1). However, risk associated with wind-farms was not regularly distributed among subpopulations. In the north, where wind-farms are very numerous (n = 497; Fig. 1a and Table 1), up to 40% of occupied territories can be considered as at risk (i.e., included within an area of 15 km radius around wind-farms; Table 1), while the central subpopulation seemed to be the least affected (ca. 5% of occupied territories under wind-farm risk; Table 1). Similarly, extinct territories within wind-farm risk zones were more frequent in the north and the south (Table 1).

3.3. Population effects of wind-farms

Taking into account previous estimates of mortality rates at wind-farms in the southern subpopulation, we calculated potential effects of these developments in the other two subpopulations (Table 2). Changes in survival rates of territorial and non-territorial birds were larger in the northern subpopulation. In the central subpopulation, wind-farms affect a low number of breeding territories (Table 1), so survival rate of territorial birds was only slightly affected (Table 2).

All population projections have stochastic growth rates (stoc-r) lower than 1 (ranges: Null models = −0.083 to −0.242; Wind-farm models = −0.091 to −0.254), indicating that the entire meta-population is decreasing. This detrimental result derived from simulations is consistent with field monitoring showing a generalized decline of the species in Spain (Carrete et al., 2007). However, wind-farms worsen the situation since mean population sizes were not always equal in the two scenarios considered (Fig. 3a). Except for the smaller core, mean population sizes were significantly larger in null models than in models including wind-farm mortality (Fig. 3, Table 3). Accordingly, models including mortality rates associated with wind-farms significantly increased probability of extinction of all subpopulations by reducing their time to extinction and, therefore, of the entire meta-population when compared to null models (Fig. 3, Table 3). Larger connectivity rates between subpopulations accentuate the effects of wind-farm mortality (Fig. 3), although it seems unlikely taking into account previous information on dispersal of the species (see before). Thus, wind-farms should be considered as having a potentially notable effect on meta-population dynamics. Regardless, GLMM showed that wind-farm effects on mean population sizes were consistent across the dispersal rates used (Table 3).

4. Discussion

Most research on the effects of wind-farms on wildlife has been based on short-term, local-scale studies devoted to quantifying collision rates of birds with turbines as well as to factors involved in influencing interspecific and local variability (for revisions, see Drewitt and Langston, 2006; Kuvlesky et al., 2007; Stewart et al., 2007). Although reviews on this subject show that the impacts of wind-farms on birds can be, in some cases, statistically significant, considerable uncertainty remains about whether the impacts are biologically significant and whether the magnitude of these impacts is substantial. Thus, the widespread belief is that wind-farms have, at most, a low impact on animal populations (*what is 3% of a bird?* Marris and Fairless, 2004), displacements of individuals to other areas without disturbance but less probably long-term reductions in population sizes being responsible for local changes in abundances. However, even when impacts were highly variable depending on species and location, longer operating times of wind-farms have been negatively related to bird abundances suggesting that short-term monitoring is not adequate to assess the actual impacts of wind-farms and, of greater concern, that negative effects of wind-farms on bird abundances could result in major impacts in only a few decades (Stewart et al., 2007).

Wind-farm impact on population viability has been largely overlooked even when low levels of additional mortality can be significant for long-lived species with low productivity and slow maturation rates, as is the case with many endangered or rare long-lived species (Sæther and Bakke, 2000). In this sense, demographic models including specific life-history traits are the only valuable tools to properly examine the actual impact of wind-farms on wildlife, with all other attempts being futile speculations lacking consistent support. Here, we present one of the few attempts to assess demographic consequences of wind-farm mortality on a long-lived endangered species, using as a study model the Spanish population of Egyptian vultures. Although not quantitatively precise, our results should be viewed as a qualitative warn-

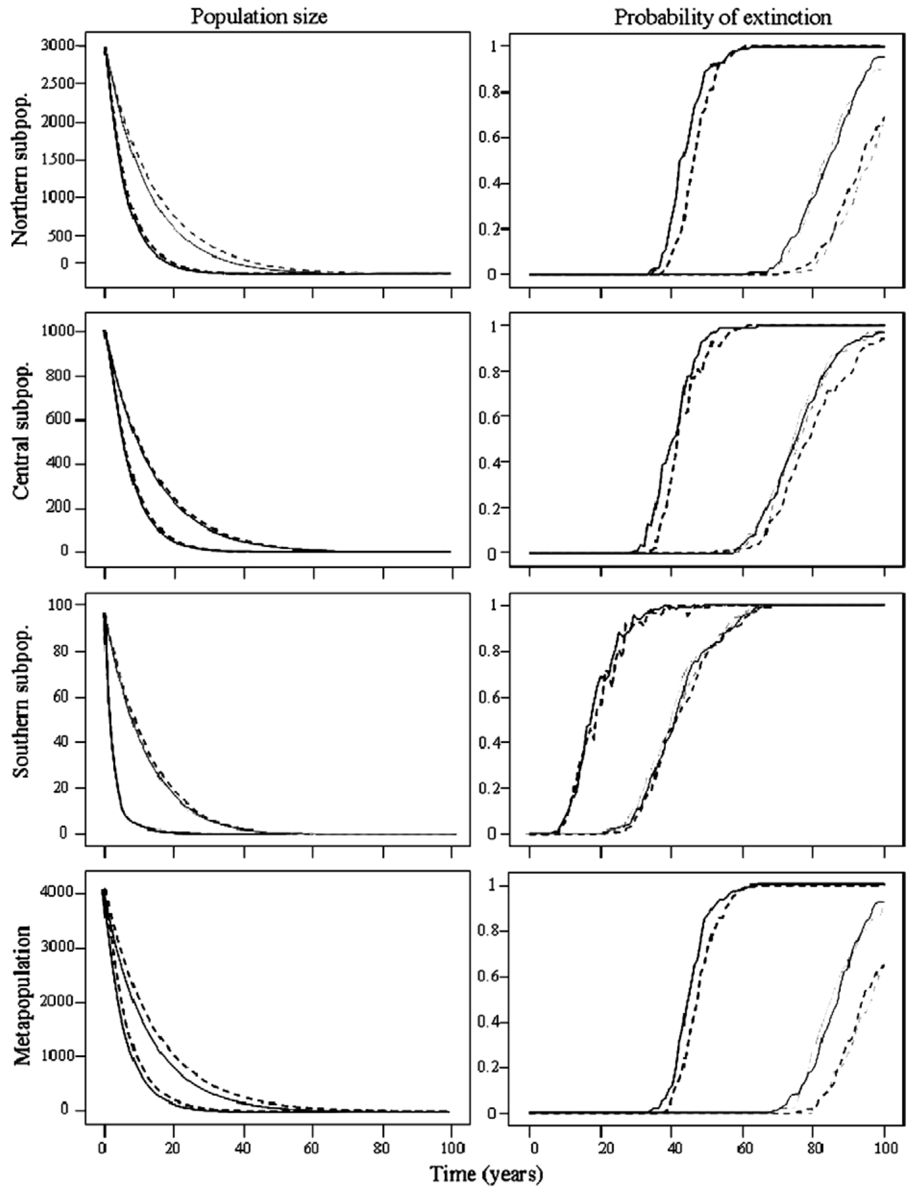

Fig. 3. Prospective trajectories of the three subpopulations and the whole meta-population of Egyptian vultures in Spain considering survival rates obtained using the basic survival model of the species (null model; dashed lines), and survival rates expected under wind-farm risk (wind-farm model; solid lines). Simulations were done under three dispersal scenarios, namely low (light grey line), medium (dark grey line), and high (black line) dispersal rates (see text for details). Lines are mean values of the stochastic runs for each time step.

ing exercise showing how very low reductions in survival rates of territorial and non-territorial birds (−0.015 and −0.008, respectively) associated with wind-farms can have significant population impacts. It should be mentioned that incorporating other aspects into modelling procedures such as detailed spatial dimensions effects would be even more severe (e.g., Nielsen et al., 2008). Regrettably, this paper does not represent a local, specific situation but a widespread scenario that should be carefully analyzed given the many other long-lived endangered species also killed at wind-farms in different European countries (e.g., at least 10 white-tailed sea eagle *Haliaeetus albicilla* per year in Norway; http://www.statkraft.com/pub/wind_power/feature_articles), in the US (e.g., 65 golden eagles *Aquila chrysaetos* in California per year; Smallwood and Thelander, 2008) and in Australia (e.g., at least 12 Tasmanian wedge-tailed eagles *Aquila audax fleayi* in 4 years; http://www.windaction.org/news/17683). Considering that these records are larger than our data on mortality and applying both scientific and ethical rationality (Shrader-Frechette and McCoy, 1992; Cooney, 2004), we alert on potential negative effects of wind-farms on the conservation of many endangered species. Moreover, taking into account the Precautionary Principle, which was recognized as a fundamental element of environmental policy at the Rio Conference of 1992, we recommend considering this cause of mortality as an important factor potentially jeopardizing wildlife conservation worldwide.

In general, territorial raptors are faithful to their breeding sites, and most studies on wind-farm effects indicate that disturbance, and, therefore, displacement of birds to other areas, appears to be negligible (Madders and Whitfield, 2006). Thus, wind-farm effects are expected to be mostly direct, through risk of collision with turbines. Although sample size is small and our search procedures may be slightly biased toward territorial birds (see methods), data recorded during our intensive monitoring suggest that birds linked to specific areas such as territorial owners, those fledging during dependence periods, or birds prospecting for vacancies in the breeding population are more prone to death by turbines. As occurs with other species (Fielding et al., 2006; Tellería, 2009a,b), the location of a wind-farm can be important in reducing their

Table 3
Effect of wind-farm mortality on population sizes and probability of extinction. Time was included in models to reduce non-independence in temporal trends while dispersal rates were considered as a random term.

	Time	Wind-farm effect	Dispersal rates
Mean population size			
North	$F_{1,601}=6746.97$, $p<0.0001$	$F_{1,601}=53.95$, $p<0.0001$	$z=1.00$, $P=0.1590$
Centre	$F_{1,601}=6199.64$, $p<0.0001$	$F_{1,601}=6.67$, $p=0.01$	$z=1.00$, $P=0.1593$
South	$F_{1,601}=1339.18$, $p<0.0001$	$F_{1,601}=0.23$, $p=0.6283$	$z=0.99$, $P=0.1616$
Meta-population	$F_{1,601}=7043.03$, $p<0.0001$	$F_{1,601}=43.01$, $p<0.0001$	$z=1.00$, $P=0.1590$
	Time	Wind-farm effect	Dispersal rates
Probability of extinction			
North	$F_{1,595}=4305.96$, $p<0.0001$	$F_{1,595}=938.13$, $p<0.0001$	$z=1.00$, $P=0.1588$
Centre	$F_{1,595}=6241.15$, $p<0.0001$	$F_{1,601}=141.69$, $p<0.0001$	$z=1.00$, $P=0.1588$
South	$F_{1,595}=554.13$, $p<0.0001$	$F_{1,601}=40.61$, $p<0.0001$	$z=0.99$, $P=0.1603$
Meta-population	$F_{1,595}=4835.97$, $p<0.0001$	$F_{1,601}=883.88$, $p<0.0001$	$z=1.00$, $P=0.1590$

impact on Egyptian vultures. The translation of our results into management guidelines is not straightforward since territory owners -and also non-territorial birds, varied in their individual patterns of movements (J.A. Donázar, J.R. Benítez, J.A. Sánchez-Zapata, M. Carrete, A. Cortés, J.M. Grande, unpublished data). However, site-specific features of territory usage apart, turbines located within a 15 km radius from nests (the equivalent to our risky zones) should be *a priori* avoided. Occupied but also vacant territories of the species should be taken into account since new colonizations or breeding dispersal events would occur mostly at these sites (Carrete et al., 2007).

Sources of extinction risk that increase mortality rates of reproductive individuals and, therefore, perturb the balance between fecundity and longevity can be particularly harmful for species with slow-lifestyles (i.e., species with large body size that mature late, produce few offspring and have a long life expectancy, Owens and Bennett, 2000), such as many raptor species. Thus, management actions should be directed toward the eradication, if possible, or reduction of factors affecting survival rate of breeders (e.g., Whitfield et al., 2004; Carrete et al., 2005; Oro et al., 2008). In our case, and despite difficulties in recording accurate information, the illegal use of poison to control predators was significantly implicated in the vacancy of numerous otherwise suitable territories (through the death of territory owners) and in the rapid population decline of the Egyptian vulture in Spain (Carrete et al., 2007). Other causes of mortality in territorial birds such as ingestion of veterinary drugs, electrocution or collision with power lines and, recently, collision with turbines at wind-farms have been also recorded (Blanco et al., 2007; J.A. Donázar, J.R. Benítez, J.A. Sánchez-Zapata, J.M. Grande, unpublished data). Even when the relative importance of all of these factors is very difficult to establish mainly because detectability of corpses is different depending on the cause of mortality, they have a cumulative effect that should be constraining population dynamics. Unlike other non-natural causes of mortality, such as illegal poisoning, which are difficult to eradicate or control, wind-farm fatalities can be easily reduced by simply powering down risky turbines during specific periods of the year at operating wind-farms showing high collision rates. Current strategies implemented by power companies are based on vigilance of risky areas such that turbines are stopped when birds approach them. However, this measure has been completely inefficient in cases of solitary territorial birds that are rarely detected by observers (J.A. Donázar, J.R. Benítez, J.A. Sánchez-Zapata, M. Lobón, unpublished data). Other more drastic solutions such as removing specific turbines or wind-farms are sometimes necessary to reduce the incidence of this mortality factor. In some areas of Spain where large numbers of bird fatalities associated with wind-farms have been reported governmental organizations are adopting these kinds of measures. However, the most effective action should be to avoid wind-farm developments in areas important for birds.

Most public and governmental support to wind-farms is based on their capacity to generate energy while not contributing to air pollution associated with fossil fuel technologies. Thus, some authors have speculated that their negative effects should be contextualized by considering negative effects associated with climate change (Stewart et al., 2007). However, the reality of the world of wind-energy is much more complex. First, wind-energy represents only a very small percentage of the total energy currently used (e.g., 1% in the US, the country with the highest contribution of greenhouse gases and the world's number one wind-power producer) and, regrettably, future projections predict an increase to no more than 6% of total energy used (American Wind Energy Association). Secondly, because wind is an intermittent resource, wind-farm production must rely on conventional power plants to back up its supply. Thus, when combined with the CO_2 emitted and pollutants released in the manufacture and maintenance of wind-farms (i.e., turbines and the associated infrastructure), substituting fossil fuels for wind-energy does little to reduce air pollution. Thirdly, and no less important, wind-farms are land-intensive and unsightly, their maintenance requires kilometres of roads and power lines which also contribute to reducing habitat availability and quality (Ferrer and Janss, 1999; Bautista et al., 2004; Sergio et al., 2004; Laiolo and Tella, 2006), and can seriously jeopardize endangered or rare bird species through collision fatalities, as shown in this paper.

5. Conclusions

The development of wind-energy is a central component of the European objective of reducing the emission of greenhouse gases by increasing the proportion of energy derived from renewable sources. Indeed, at the end of 2008, there are more than 640 wind-farms in Spain capable of producing 14,145 MW, and the objective envisaged in the 5-year Spanish Plan for Renewable Energies is to reach 20,155 MW in 2010 (Spanish Wind Energy Association, http://www.aeeolica.org). Thus, the expansion of wind-farms in Spain is set to continue. However, as we have shown here, current wind-energy developments have a real impact on globally endangered wildlife, and the potential implications of wind-farms for birds (and although less studied, for bats) are of even greater concern when considering the scale of future proposals. Immediate solutions to this conservation problem must involve powering down or removing risky turbines or wind-farms, while future locations should be planned with consideration of the spatial distribution of endangered or rare bird species. In the long-term, alternative solutions to present wind-farm developments should be considered in order to reconcile biodiversity conservation to human development. Meanwhile, as we shown here and as a precautionary guideline, turbines within specific radius from nest sites of sensitive, endangered or rare species should be considered as risky ones.

Virtually every aspect of biodiversity from species numbers to threats to populations (Thompson et al., 2001) or sustainable harvest levels (Curtis and Vincent, 2008) are plagued with uncertainty. Thus, biodiversity conservation may be a field in which the precautionary principle is immediately and urgently relevant (Myers, 1993). However, traditional approaches have been to require evidence of environmental harm before acting to restrict individual, corporate or state actions (Cooney, 2004). Here we offer evidence against the current assumption of wind-farms have a low impact on wildlife (de Lucas et al., 2008), and we advocate the consideration of the precautionary principle when allowing this and similar human developments which can be harmful for the long-term conservation of endangered, long-lived vertebrates.

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