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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 datasets 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 15,154 MW 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 population 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