Mortality at wind-farms is positively related to large-scale distribution and aggregation in griffon vultures

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Wind-farms have negative impacts on the environment, mainly through habitat destruction and bird mortality, making it urgent to design predictive tools to use in landscape planning. A frequent assumption of wind-farm assessment studies is that bird distribution and abundance and bird mortality through collision with turbines are closely related. However, previous results are contradictory and question the usefulness of these variables to select safer wind-farm locations. We focused on a species highly vulnerable to collision at wind-farms, the griffon vulture, to test whether mortality at turbines was related to the relative position of turbines within the vulture population. We used the location of all turbines on 34 wind-farms in southern Spain, details on 342 griffon vultures found dead, and the location and size of breeding colonies and roost sites of the species during the breeding and non-breeding seasons, respectively. Using variables that describe the large-scale distribution and aggregation of vultures, we found that year-round mortality at turbines increased when they were located in highly populated areas, a result that can be translated into management guidelines to plan wind-farm locations.

Bird abundances can help to guide wind-farm plans at large scales. Current protocols of counting birds at specific points during particular periods of time have low predictive power. However, other more integrative cues such as the spatial distribution and aggregation of some vulnerable species should be used as criteria for large-scale environmental planning. Local inspection of the relationship between mortality at existing turbines and their relative position within the spatial distribution of bird populations can guide managers in planning future wind-farms and in managing currently operating developments.

1. Introduction

Wind-farms have received public and governmental support as alternative energy sources that do not contribute to air pollution as associated with fossil fuel technologies (Leddy et al., 1999). However, the expansion of wind power has environmental impacts (i.e., habitat removal, construction of roads and power lines, visual impact; Laiolo and Tella, 2006; Kuvlesky et al., 2007) that need to be evaluated and considered. In this sense, the primary emphasis of the majority of wind farm-wildlife research has been devoted to investigate how wind farm development has impacted bird and bat populations (e.g., Langston and Pullan, 2003; Baerwald et al., 2008; Garvin et al., 2011), in particular collision rates of birds with turbines as well as factors influencing interspecific and local variability (reviewed in Drewitt and Langston (2006,

2008), Kuvlesky et al. (2007), and Stewart et al. (2007)). Both aspects are central in decision-making for managers and policy makers dealing with landscape and environmental planning. An important point in the discussion of potential wind-farm effects on wildlife is the idea that collision mortality increases with bird abundance (Musters et al., 1996; Osborn et al., 2000; Drewitt and Langston, 2006; Tellería, 2009a,b). Although linearity in this relationship could be a priori a simplistic assumption because of interspecific differences in susceptibility to this infrastructure, higher abundance of individuals of species sensitive to collision at wind-farms would increase fatality rates. Scientific evidence on this subject is not conclusive, but environmentalists and managers have used this argument as a precautionary measure against the installation of wind-farms in areas with high densities of sensitive birds such as vultures and other large raptors (e.g., Kingsley and Whittam, 2005; Atienza et al., 2008).

Recently, de Lucas et al. (2008) offered a result that challenges this frequent assumption of wind-farm assessment, since bird

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abundance and bird mortality through collision with wind turbines were not closely related in their study. Authors surveyed the relative abundance (i.e., the number of birds that crossed turbine rows within 250 m of a turbine, averaged per season and year) of 10 bird species (nine raptors and one stork) during four periods (pre-breeding, breeding, post-breeding, and winter) in 2000–2002 at two wind-farms in Tarifa, southern Spain, to relate bird fatality rates recorded from 1993 to 2003 to these abundances. Among all species studied, the griffon vulture was the species most frequently killed by wind turbine collision, so authors used information on seasonal changes in its abundance and mortality (the larger number of dead birds in winter from 1993 to 2003 vs their higher abundance during winters of 2000–2001 and 2001–2002) to show a lack of relationship between mortality rate and bird abundance. Generalizing their results, authors did not find a correlation between species' mortality and their abundances at both wind-farms studied. Barrios and Rodríguez (2004), however, found that high abundances of individuals and, in particular, large numbers passing within 5 m of the blades of an operating wind turbine (which the authors considered as a 'risk situation') were the main determinants of high mortality rates at the same two wind-farms. Indeed, most dead griffon vultures were found at the wind-farm where the number of vultures observed and the overall risk of passing near turbines was higher (spatial pattern of mortality), and when vultures were more abundant and wind conditions forced them to pass more frequently near turbines (i.e., in winter). Unlike de Lucas et al. (2008), data gathered by Barrios and Rodríguez (2004) on vulture abundances suggested that birds at risk and mortality corresponded to the same study period, thus strengthening their results. However, contradictions between these studies should be resolved in order to correctly guide the installation of future wind-farms in this and other areas also important for bird conservation. Moreover, to increase our ability to estimate expected mortality instead of to simply measure it, it is also important to develop predictive tools to guide landscape planning, and in this way reduce bird mortality at wind-farms. Currently, risk assessment studies performed before wind-farm construction include the distance to different bird nest sites, including colonies of griffon vultures, as a variable to predict the potential risk of a development (Ferrer et al., 2011). However, the final evaluation encompasses all information applied to a subjective assessment, without using any objective, or standardized criteria. The high mortality rates recorded for species such as the griffon vulture at some of these approved wind-farms clearly show that some of these measures have not been well designed.

Here, we tested the key assumption that turbines located in areas with larger abundances of individuals of species sensitive to collision kill more birds than turbines located in areas with lower local abundances of the same species. Distance to colonies can also be a descriptor of mortality (Atienza et al., 2008), although variability in colony size could be important to explain differences in mortality rates among turbines. Thus, our main prediction is that large aggregations of griffon vultures increase probabilities of collision at turbines, resulting in more birds found dead at some specific turbines located within high aggregation areas. To test this prediction, we used information on dead griffon vultures gathered from the same area as the previous studies but covering a larger number of wind-farms (34 vs 2), and using precise data on the spatial distribution (i.e., distance between breeding colonies and/or roost sites and turbines) and abundance (measured through an aggregation index, see below) of the species (Fig. 1). Finally, we provide a management criterion to assess potential risk before wind-farm construction using griffon vultures as an example of a species sensitive to these developments.

2. Methods

2.1. Study area and species

Data used to perform this study were gathered in Campo de Gibraltar and Tarifa, southern Spain (Fig. 1). The area includes a series of mountains up to 840 m above sea level running in a north–south direction. There, turbines of wind-farms are arranged in rows along the ridges of mountains and hills to optimize the harnessing of energy from prevailing east–west winds. The vegetation includes natural open forests of *Quercus* sp. and shrubs interspersed with agricultural lands used for cattle grazing (for more details, see Barrios and Rodríguez, 2004; de Lucas et al., 2008).

This area is home to many resident species but is also the main European flyway for migratory birds, as well as being an important location for wind-energy production in Europe (by 2010, Cádiz represents ca. 2% of the cumulative installed capacity of the EU; <http://www.ewea.org>). Thus, some studies have been carried out to assess wind-farm effects on wildlife by using long-term mortality data taken at the two oldest developments (Barrios and Rodríguez, 2004; de Lucas et al., 2008). The main results of these studies show that most deaths are concentrated in space and time, and include a taxonomic and a migration component (a few species suffered the most losses, with species with resident populations being killed more often than migrating species). Therefore, bird vulnerability at some points at these wind-farms seems to be the resultant of a combination of site-specific (wind-relief interaction at some turbines), species-specific, and seasonal factors.

Our investigation focuses on the species most vulnerable to collision at wind-farms, the griffon vulture. Indeed, more than 50% of all birds found killed at monitored wind-farms during our study period belong to this species (see [Supplementary material](#)). Vultures are more likely to collide with turbine blades than many other avian species due to their large size (ca. 10,500 g), and thus reduced manoeuvrability, flight type (i.e., soaring-gliding) and foraging behavior (i.e., search for carcasses in large geographic areas) (Baisner et al., 2010). This long-lived, cliff-nesting scavenger breeds colonially, forming large aggregations of individuals (communal roost) during winter. The global population status of this species has not been quantified accurately, but the species is not believed to approach the thresholds for the population decline criterion of the IUCN Red List. The current status of the species is therefore evaluated as "Least Concern" (BirdLife International, 2011). In the study area, there is an important sedentary breeding population of griffon vultures (more than 1900 pairs censused in 2008) and a minimum of 4000 birds during winter months (Fig. 1; Natural History Society of Cádiz, unpublished results, 2009). Although breeding colonies and the non-breeding population (i.e., all birds during the non-breeding season) have increased in numbers during recent years, their spatial patterns of relative abundance have remained unchanged (Pearson correlation between colony sizes in 1998 and 2008: $r = 0.98$, 95% CI = 0.96–0.98, $n = 112$), and aggregations of birds (i.e., breeding colonies or communal roosts) have remained the same since the beginning of the 1980s (Junco and Barcell, 1997; del Moral, 2009; Natural History Society of Cádiz, unpublished results).

2.2. Database and explanatory variables

The complete database we are using includes: (1) the exact location (in UTM units) of all turbines ($n = 799$) at the 34 wind-farms in the study area (public information provided by the Environmental Department of the Junta de Andalucía; [Supporting information](#)), (2) details on 342 griffon vultures found killed by turbines (i.e., birds with lacerations, wing injuries, head injuries, back injuries and signs of internal injuries which were certainly

Fig. 1. (a) Distribution of griffon vultures during the breeding and the non-breeding season, and location of wind-farms in the province of Cádiz, south Spain. Breeding colonies/roost sites are represented with differently sized squares according to number of breeding pairs/individuals. (b) Detail of the area with highest density of wind-farms. Turbines are represented with differently sized dots according to mortality.

caused by a collision) between January 1998 and March 2008, and (3) the spatial distribution and size of breeding colonies and roost sites of griffon vultures during the breeding (February–July) and non-breeding (August–January) seasons, respectively (Fig. 1).

Information on dead vultures, provided by the Environmental Department of the Junta de Andalucía, includes the turbines where birds were found killed, such that a minimum mortality value can be assigned to each. As occurs in the study of [de Lucas et al. \(2008\)](#), which used a sub-sample of the database we are using, information on mortality may contain some flaws and constraints such as the lack of standardized intervals for searches of mortalities at turbines at some wind-farms. As carcasses remain in the field for a period of time longer than the time between successive searches (at least twice a week in all wind-farms, see text) we believe that this source of variability does not seriously bias our results. However, to reduce its potential effects, we have included as a fixed factor the existence of an intensive survey protocol associated with wind-farms with preventive vigilance, consisting on an intensive daily survey of the wind-farm to stop particular turbines to avoid collisions of birds flying in their surrounding (in 24 out of the 34 wind-farms surveyed). In an experiment performed by [Barrios and Rodríguez \(2004\)](#) where authors followed the time to disappearance of eight griffon vultures with known dates of death, they found that all these carcasses persisted within the sampling area for several months. These results are in accordance with other ones showing that small birds disappear earlier and at a higher proportion than larger birds ([Smallwood, 2007](#); [Ponce et al., 2010](#)). Thus, as other authors working in the same area (e.g., [Barrios and](#)

[Rodríguez, 2004](#); [de Lucas et al., 2008](#)), we did not correct for carcass removal by predators and scavengers, this effect being negligible when dealing with large species and monitoring wind-farms at least twice a week as in our study area.

The distribution and size of all breeding colonies of the species were obtained through simultaneous census (i.e., all colonies counted at the time) performed by experienced local ornithologists during February–March of 1998 and 2008 ([del Moral, 2009](#)). All cliffs known to have been occupied by the species as well as potential suitable sites were visited during the mating, incubation and nestling periods to record their occupancy and the number of breeding pairs through standardized protocols usually applied to this species ([del Moral, 2009](#)). During January of 2008 and 2009, the non-breeding population was also monitored by counting (at dusk) individuals in all breeding colonies, which in winter are used as communal roosts, and by observing other potentially suitable roosting sites. In 2008, the griffon vulture population of the study area comprised ca. 2000 breeding pairs distributed in 122 colonies (ranging from 1 to 209 breeding pairs). A minimum of 3800 individuals was counted in the winter while roosting in 60 communal roosts (Fig. 1).

To test whether mortality at turbines was related to the relative position of turbines within the griffon vulture population, we obtained a bird aggregation index (Aggregation) for each turbine for the breeding and the non-breeding seasons using information on griffon vulture distribution (location and mean size of the breeding colonies and communal roosts obtained after averaging breeding censuses of 1998 and 2008, and wintering censuses of 2008 and

2009; del Moral, 2009; Natural History Society of Cádiz, unpublished results). This index was calculated following Moilanen and Hanski (1998) as $\text{Aggregation}_i = \exp(-d_{ij})A_j$ where d is distance, i is the i th turbine, j is j th breeding colony or roost site (depending on the season), and A_j is the size (in number of breeding pairs or individuals) of the j th breeding colony or roost site, for the breeding and the non-breeding seasons, respectively. This index actually describes connectivity (sensu Moilanen and Hanski, 1998), larger values representing higher spatial aggregations of the species around each turbine. We also included distance to the nearest breeding colony or roost site of the species in models, depending on the season, because of the usefulness of this variable for management purposes. We considered whether bird mortality throughout the year could be predicted with information on the breeding period, using information of turbine distances to the nearest breeding colony and their aggregation in this period. Distance to breeding colonies is a criterion usually incorporated in risk assessment studies to evaluate the suitability of a site for wind-farm construction (Ferrer et al., 2011). Several authors have proposed that turbine attributes may increase collision risk (e.g., Barrios and Rodríguez, 2004; de Lucas et al., 2008). Therefore, we also used information on turbine height and altitude above sea levels as potential predictors of mortality in models.

2.3. Analytical procedures

We used Generalized Linear Mixed Models (McCullagh and Searle, 2000) to construct predictive models of bird mortality through the year. We constructed a basic model including variables not related to bird distribution but previously linked to mortality at wind-farms, namely: time since monitoring (the accumulative number of birds killed increases with the number of years a wind-farm is monitored), season (more birds are killed in winter), altitude above sea level and height of turbines (turbine characteristics affect bird mortality) (e.g., Barrios and Rodríguez, 2004; de Lucas et al., 2008). Then, we tested whether variables describing bird abundance near turbines (i.e., aggregation of birds at breeding colonies or roost sites and distance to the nearest breeding colony or roost site) improved the statistical power of the basic model. We modelled number of bird deaths per turbine using the logarithmic link function and the Poisson error distribution. A previous study supports that mortality is spatially aggregated at just a few turbines within wind-farms (Barrios and Rodríguez, 2004). Indeed, a preliminary analysis of the database we are using showed that wind-farms killed griffon vultures differentially, with most deaths (52%) occurring at two developments. Moreover, compared with a random pattern of mortality (i.e., equal mortality rates across turbines), observed values of dead birds at turbines were spatially aggregated in both seasons (breeding season: $\chi^2 = 2385.75$, $p < 0.001$, $df = 4$; non-breeding season: $\chi^2 = 2854.89$, $p < 0.001$, $df = 6$), with all dead birds found at 89 turbines during the breeding season (ca. 11% of 799 turbines) and 164 turbines during the non-breeding season (ca. 21% of 799 turbines). Thus, we considered turbines as our sampling units, and they were included (nested within wind-farms) as a random term to control for potential, unmeasured differences among them such as topography or location, while also avoiding non-independence in mortality data.

Models were obtained using the GLIMMIX procedure in SAS 9.2 (SAS Institute Inc., 2009), and their relative explanatory power was compared, penalizing for complexity, using differences in AIC scores (lower scores indicated greater statistical support; Burnham and Anderson, 2002; Richards, 2005). Models with AIC scores differing from that of the lowest score by more than two (i.e., DAIC) were considered to be unsupported statistically (Richards, 2005).

3. Results

3.1. Distribution, aggregation and mortality of griffon vultures at turbines

The inclusion of a variable describing bird aggregation during the breeding season greatly improved the statistical power of the basic model just including variables linked to wind-farm monitoring and turbine characteristics (Table 1). Indeed, these models (whose difference is the inclusion or not of turbine altitude) are 4.76 (model 1: 1/0.21) and 3.33 (model 2: 0.7/0.21) more likely, given the data, than the basic one (Table 1). The other models including different descriptors of vulture abundance (year-round aggregation of birds, distance to breeding colonies and/or roost sites) had less statistical support (Table 1). These results show how turbines located in areas with higher aggregations of the species (i.e., large breeding colonies) killed more birds than turbines located in less populated areas. Thus, although other non-evaluated factors are probably still affecting mortality at specific turbines, their location within the breeding vulture population is related to the number of bird deaths, even when controlling for time since wind-farm operation, for the existence of different intensities in search procedures (i.e., wind-farms with or without preventive vigilance) and for turbine characteristics (height and altitude above sea level). It should be noted that although more birds died during the non-breeding season, the main variable to describe this pattern is aggregation during the breeding period. Similarities in the aggregation pattern of the species during the breeding and the non-breeding seasons (large colonies remain as large communal roosts but small colonies disappeared; correlation between colony and roost site sizes: $r = 0.70$, 95% CI = 0.48–0.83, $n = 39$) can explain this result.

3.2. Security distance: application to wind-farm planning

To guide managers in wind-farm planning, we modelled the relationship between the relative positions of turbines within the spatial distribution of the population of griffon vultures combined with data on mortality (obtained as presence/absence of death birds per turbine; error distribution: binomial, link function: logit) to obtain a setback distance for the installation of turbines. The best model included information on aggregation within breeding colonies, time since monitoring, existence of preventive vigilance protocols and season. The alternative model was similar, also including turbine altitude (DAIC = 0.21). The other models had less statistical support (DAIC > 2.01, Table 2). Expected probability of mortality could be thus presented against aggregation of breeding colonies (see Fig. 2 as an example using our local data), and managers can select from this plot the maximum year-round probability of mortality that they would assume when approving a wind-farm placement. This value represents an objective criterion developed by taking information on the study area that could be applied during risk assessments.

Similar procedures could be applied to data gathered in other areas to adjust security placement to local variability in mortality. Moreover, this method can be also validated using species-specific information on mortality and aggregation of other species of interest. In the case of territorial species, distance to territories as well as aggregation (computed by using a modified version proposed by Carrete et al. (2006)) should be considered, as differences in social systems can change their relative importance as mortality predictors. As the limits imposed to wind-farm locations can be changed depending on the risk that could be assumed (which also depend on the species considered), this seems a straightforward and applicable criterion to be used in many different regions, situations and to a variety of species of conservation concern.

Table 1

Comparison of models (using Akaike information criterion, AIC) to predict the effects of location of griffon vultures around wind-farm turbines (aggreg BC: vulture aggregation during the breeding season, aggreg BC/RS: vulture aggregation during the breeding and non-breeding season, dist BC: distance to the nearest breeding colony, dist BC/RS: distance to the nearest breeding colony or roost site) on the number of birds found dead at wind farms in the area of Campo de Gibraltar and Tarifa, Cádiz. Smaller AIC values suggest a better fit of the model to data. Model likelihoods represent the relative likelihood of a model and were calculated by comparing the AICw (Akaike weight) of each model to that of the best one ($AICw/AICw_{\text{best model}}$). In bold, models with $DAIC < 2$ (alternative models). Models with $DAIC > 10$ are not shown. Time: time since monitoring, season: breeding and non-breeding seasons, vigilance: existence of preventive vigilance in the wind-farm, height: turbine height; altitude: turbine altitude above sea level.

Model		AIC	DAIC	AIC _w	Model likelihood
1	Aggreg BC, time, season, vigilance, height	1534.69	0	0.30	1.0
2	Aggreg BC, time, season, vigilance, height, altitude	1535.48	0.79	0.20	0.7
3	Aggreg BC/RS, time, season, vigilance, height	1537.23	2.54	0.08	0.3
4	Aggreg BC, aggreg BC/RS, time, vigilance, height, altitude	1537.37	2.68	0.08	0.3
5	Aggreg BC, dist BC, time, season, vigilance, height, altitude	1537.43	2.74	0.08	0.3
6	Time, season, vigilance, height, altitude	1537.85	3.16	0.06	0.2
7	Aggreg BC/RS, time, season, vigilance, height, altitude	1538.34	3.65	0.05	0.2
8	Dist BC, time, season, vigilance, height	1538.36	3.67	0.05	0.2
9	Dist BC/RS, time, season, vigilance, height	1539.06	4.37	0.03	0.1
10	Dist BC, time, season, vigilance, height, altitude	1539.29	4.6	0.03	0.1
11	Dist BC/RS, time, season, vigilance, height, altitude	1539.65	4.96	0.03	0.1
12	Dist BC, Distance BC/RS, time, season, vigilance, height, altitude	1541.26	6.57	0.01	0.0

Table 2

Comparison of models (using Akaike information criterion, AIC) linking the relative positions of turbines within the spatial distribution of the breeding population of griffon vultures and mortality data (aggreg BC: vulture aggregation during the breeding season, aggreg BC/RS: vulture aggregation during the breeding and non-breeding season, dist BC: distance to the nearest breeding colony, dist BC/RS: distance to the nearest breeding colony or roost site) at wind farms in the area of Campo de Gibraltar and Tarifa, Cádiz. Smaller AIC values suggest a better fit of the model to data. Model likelihoods represent the relative likelihood of a model and were calculated by comparing the AICw (Akaike weight) of each model to that of the best one ($AICw/AICw_{\text{best model}}$). In bold, models with $DAIC < 2$ (alternative models). Time: time since monitoring, season: breeding and non-breeding seasons, vigilance: existence of preventive vigilance in the wind-farm, height: turbine height; altitude: turbine altitude above sea level.

Model		AIC	DAIC	AIC _w	Model likelihood
1	Aggreg BC, time, season, vigilance, height	1143.51	0	0.28	1.0
2	Aggreg BC, time, season, vigilance, height, altitude	1143.72	0.21	0.25	0.9
3	Time, season, vigilance, height, altitude	1145.52	2.01	0.10	0.4
4	Aggreg BC/RS, time, season, vigilance, height	1146.03	2.52	0.08	0.3
5	Dist BC, time, season, vigilance, height	1146.34	2.83	0.07	0.2
6	Aggreg BC/RS, time, season, vigilance, height, altitude	1146.47	2.96	0.06	0.2
7	Dist BC, time, season, vigilance, height, altitude	1146.5	2.99	0.06	0.2
8	Dist BC/RS, time, season, vigilance, height, altitude	1147.22	3.71	0.04	0.2
9	Dist BC/RS, time, season, vigilance, height	1147.6	4.09	0.04	0.1
10	Dist BC, dist BC/RS, time, season, vigilance, height, altitude	1148.29	4.78	0.03	0.1

Fig. 2. Relationship between aggregation of vultures around turbines and mortality risk during the breeding (black line) and non-breeding (gray line) seasons. The dashed line shows a maximum aggregation value (~4.5) above which wind-farms should not be installed (security distance) to reduce mortality risk of this species to 0.5.

4. Discussion

The assumption that bird mortality at wind-farms is related to local bird abundances has been generally assumed as a positive relationship (Musters et al., 1996; Osborn et al., 2000; Drewitt and Langston, 2006; Kuvlesky et al., 2007; Tellería, 2009a, 2009b; Bright et al., 2008) but has been scarcely evaluated. Interestingly,

evaluation of this relationship has shown somewhat contradictory results (e.g., Barrios and Rodríguez, 2004; Everaert and Steinen, 2007; de Lucas et al., 2008). Here, we present tested support to the positive relationship between the relative abundance of griffon vultures, a species sensitive to collision at wind-farms (see below), and their mortality at wind-farm turbines. This finding contradicts previous results obtained in the same study area by de Lucas et al. (2008), but supports those of Barrios and Rodríguez (2004). The apparent lack of strength in this relationship between studies could be a consequence of sampling procedures. De Lucas et al. (2008), for example, based their main conclusion on two points, namely: (1) a lack of correlation between mortality rates obtained in a 9-year period and local (i.e., within wind-farms) bird abundances from only 1 year at two wind-farms, and (2) comparison between the abundance and mortality of griffon vultures recorded among seasons ($n = 4$). Conversely, Barrios and Rodríguez (2004), studying the same wind-farms but considering concurrent mortality rates and vulture abundances, found that more birds died when more birds were present at these developments. Moreover, their mortality rate increased during winter, the season during which bird flight behavior is riskier (i.e., higher ratio of birds observed within 5 m of the blades of operating turbines out of the total number of passes or observations within 250 m of the turbine lines; Barrios and Rodríguez, 2004). Overlooking inter-annual differences in local abundances of vultures and seasonal changes in bird flying behavior is a potentially serious problem that may affect for example avoidance behaviors (i.e., behaviors precluding risky situations such as flying in the vicinity of rotors; Chamberlain et al., 2006). In

the first case, changes in food abundance and distribution as well as in meteorological conditions can affect movements and, therefore, the number of birds flying at specific points from 1 year to the other (Martínez-Abraín et al., in press). In the second case, seasonal variations in bird flying behavior change the risk of collision of individuals throughout the year and are a major determinant of mortality rates (Barrios and Rodríguez, 2004; Garthe and Hüppop, 2004; Hoover and Morrison, 2005; Chamberlain et al., 2006). Thus, a coarser estimation of the abundance of individuals of a species potentially using the area (such as that used here) could be a better predictor of actual mortality than a finer but more punctual and local estimation subject to considerable observer, stochastic and systematic errors that affect its accuracy and precision (Chamberlain et al., 2006).

Regarding our previous comments, it seems that counting birds at specific points during particular periods of time has a very low predictive power when planning future developments. However, the use of other estimates of bird abundances such as the large-scale spatial distribution of breeding colonies as well as their sizes could be more promising, mainly when dealing with species faithful to their territories, colonies or wintering areas such as many raptors. In these cases, even when birds can change their daily movements in response to local variability in resources (i.e. food) or conditions (i.e. wind), they must travel around their colonies or roost sites (Carrete and Donazar, 2005), and must face nearby turbines into which they may collide. Using this latter approach, we corroborated the positive relationship between bird distribution (location and size of bird aggregations), and mortality at wind-farm turbines, while offering a predictive tool useful in forecasting future impacts before the installation of new developments. Our results indicate that precise information on the distribution and abundance of sensitive species, such as large raptors, might be the best biological guideline to assess wind-farm location at a large scale (e.g., Tellería, 2009b; Bright et al., 2008). An interesting result is that aggregation of birds around turbines, but not their distance to breeding colonies, explain annual mortality. Thus, even when risk assessment considers distance to breeding colonies as a factor to evaluate the possibility of constructing a wind-farm, the combination between location and abundance of birds should be taken into account as a primary factor. Importantly, as our results indicate, the importance of aggregation of birds around turbines can be used to evaluate wind-farm locations in terms of the collision risk of susceptible species.

Previous studies have indicated a spatial overlap between wind-farms and species of conservation concern (e.g., Fielding et al., 2006; Tellería, 2009a, 2009b; Carrete et al., 2009), while others have provided sensitivity maps based on bird distributions to guide wind-farm installation (e.g., Bright et al., 2008). These proposals are commonly viewed as risk maps where wind-farm impacts should be minimized. However, interpretations of these maps should not mistake interspecific differences in sensitivity to collision for conservation concern. The mortality database we are using shows how, compared to other raptor species, griffon vultures are particularly vulnerable to collision (Martínez-Abraín et al., in press). Thus, large regional abundances of this species can increase mortality rates to values as great as ca. 400 birds/year in areas such as Navarre (north Spain; Drewitt and Langston, 2006). In the other extreme, however, even very low values of additional mortality rates recorded at wind-farms can have negative demographic consequences on species of conservation concern (Martínez-Abraín et al., in press). Recently, Carrete et al. (2009) show how population sizes and therefore time to extinction of an endangered long-lived raptor, the Egyptian vulture *Neophron percnopterus*, strongly decreased when an additional wind-farm annual mortality rate of 0.015 and 0.008 for territorial and non-territorial birds, respectively, is included in population viability models. Thus, to assume

that collision mortality should increase with bird abundance because more birds are 'available' for collision may be too simplistic. Perhaps just as simplistic as to make interspecific generalizations about this relationship without taking into account specific sensitivities to collision at turbines. Hence, although landscape planning should include the spatial distribution and aggregation of sensitive species, other aspects such as conservation status (Garthe and Hüppop, 2004) should also be taken into account when selecting future wind-farm locations as well as when managing currently operating developments.

When a wind project is proposed in European countries, an environmental impact assessment is required by environmental authorities. These studies must include a section assessing the impact that the development is likely to have on the site's bird populations (Environmental Impact Assessment Directive 97/11/EC). Environmental authorities use the overall assessment to reach a declaration on the environmental impact stating the significance and acceptability of the predicted effects. These declarations identify additional measures to mitigate and compensate potential negative environmental consequences and other conditions that should be met by the project developer such as the monitoring of the environmental impacts. If properly enforced, the environmental impacts hierarchy "avoid-minimize-compensate" would provide the regulated community with incentives to prevent wildlife and habitat impacts in sensitive areas and, if necessary, compensate for residual impacts through restoration or conservation projects (Cole, 2011). In Spain, wind-farm planning is competency of regional governments. In our case, the planning of wind-farm projects corresponds to the Andalusian government. Here, the project's location is evaluated considering spatial environmental data available in the area (i.e., statutory protected areas, Important Bird Areas, nest sites, roost sites, rubbish dumps, pre-migratory settlement areas, eagle dispersal areas). A large amount of digitized data is available in Andalusia and rarely should a breeding site of an endangered bird species be overlooked in this phase of planning. Risk areas are established around breeding sites using a "risk radius" based on available data of species' home ranges (Janss et al., 2010). However, risk radii have been arbitrary or based on old-fashioned data (Carrete et al., 2010). Consequently, observed mortality can be sometimes higher than that previously expected (Carrete et al., 2010). Monitoring program of environmental impacts imposed on approved projects are thus an important complement whose results should be used to implement corrective measures such as powering down or removing risky turbines and/or farms, and by placing them outside areas critical for endangered birds. It is relevant to consider that, for example, in our study area all vultures were killed by 214 out of 799 turbines (ca. 27%) and more than 50% of those deaths occurred in only two wind-farms. Nevertheless, the current framework is proving ineffective for planning, and the reality is that removing or reforming existing wind-farms is virtually impossible due to the absence of regulatory and flexible administrative mechanisms (only the activity of three wind-farms in Spain in Navarre, Castilla-León and Valencia have been paralyzed because of high mortality rates of birds).

Making decisions that affect biodiversity conservation has often to be made on the basis of incomplete information, ignorance about processes and speculations about outcomes. In this sense, it is important to anticipate and take action to avert potential, uncertain environmental harm. The precautionary principle is a widely and increasingly accepted general principle of environmental policy, law, and management (Cooney, 2004). In the wind-farms-wildlife scenario, we reaffirm the need for applying the precautionary principle during wind-farm planning to minimize the impact of wind-farms on populations of long-lived species (Carrete et al., 2010).

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at [doi:10.1016/j.biocon.2011.10.017](https://doi.org/10.1016/j.biocon.2011.10.017).

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