Wind turbine fatalities approach a level of concern in a raptor population

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ABSTRACT

Mortality from collisions with increasing numbers of wind turbines is a potential hazard to raptor populations, but the actual effects on a population scale have rarely been studied based on field data. We estimated annual collision numbers for Red Kites *Milvus milvus* in the German federal state of Brandenburg (29,483 km^2). A hierarchical model considering carcass persistence rate, searcher efficiency and the probability that a killed animal falls into a searched area was applied to results of carcass searches at 617 turbines. Collision risk varied significantly with season. The model estimated 308 (95% CrI 159–488) Red Kite fatalities at 3044 turbines operating during 2012, representing 3.1% of the estimated post-breeding population of 9972 individuals. Using the potential biological removal (PBR) method, mortality thresholds of 4.0% were obtained for migratory Red Kite populations. This level of mortality may be reached when turbine numbers increase within a few years. Since wind turbine collisions may affect Red Kites throughout the global range, a more detailed assessment of the actual impacts on populations is needed, especially because the PBR does not account for the predominance of adult birds among the collision victims.

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Introduction

The use of wind turbines to generate renewable energy has rapidly increased in the Northern Hemisphere during the last 15 years (Pullen & Sawyer 2011). The resulting increase in the number of wind farms has raised concern about their potentially negative, cumulative effects on bird populations (Drewitt & Langston 2006), but empirical evidence is scarce. Raptors *Accipitriformes* are regarded as particularly vulnerable, either to displacement from habitats, or to increased mortality due to collisions at turbines (Madders & Whitfield 2006). The two processes of displacement and collision are to some extent mutually exclusive, since birds avoiding the need to approach a wind farm would not collide with the turbines. Collision mortality occurs in various raptor species and throughout the annual cycle (Barrios & Rodriguez 2004; Dahl et al. 2012; Smallwood & Thelander 2008), hence it is probably the greater risk for many species. Raptors are long-lived species with relatively low reproductive rates, and their populations may be especially sensitive to increases in adult mortality (Saether & Bakke 2000). Where wind farms occupy large parts of a species' breeding

range (e.g., Tellería 2009), large scale population effects may be suspected. At this scale, however, the cumulative total numbers of fatalities at multiple wind farms are poorly known. Field studies are usually restricted to local or regional populations (Carrete et al. 2012; Dahl et al. 2012). Studies assessing impacts on larger populations therefore used collision risk models instead of mortality estimates from field data (Carrete et al. 2009; Eichhorn et al. 2012; Schaub 2012).

The Red Kite *Milvus milvus* is an endemic raptor species widespread across temperate and southern Europe. Regional populations are declining in most of its breeding range and the species is regarded as "near threatened" (Knott et al. 2009). In addition to direct killing, accidental poisoning and electrocution, high numbers of reported fatalities at wind turbines have recently been identified as a major source of additional mortality (Knott et al. 2009; Langgemach et al. 2010). Red Kites search for prey in open landscapes. Especially in summer, when crops become too tall and dense for foraging, they may be attracted to wind farms by favourable hunting conditions around the tower. Because most search flights are performed at rotor height they are especially vulnerable to collisions at wind turbines (Mammen et al. 2011).

Environmental impact assessments in most countries focus on the potential effects of a single wind farm (Ferrer et al. 2012). Because collision risk with each turbine is small, such assessments will usually fail to predict negative effects on a population scale. Models predict negative effects on populations if large numbers

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Table 1
Post-breeding population structure under a stable age distribution, given as proportion of the total population.

Age	Annual survival [a]	Breeding probability[b]	Proportion of individuals	Proportion of breeding individuals
0	0.60	0	0.30	0
1	0.74	0	0.19	0
2	0.84	0.20	0.13	0.03
3	0.84	0.60	0.09	0.05
4	0.88	0.80	0.06	0.05
5	0.90	0.98	0.23	0.23
Total			1.00	0.36

[a] Schönfeld (1984), Nachtigall (2008).
[b] Values chosen after Newton et al. (1989) and Nachtigall (2008).

of wind farms are placed in Red Kite home ranges (Eichhorn et al. 2012; Schaub 2012), but these effects have not yet been demonstrated for real populations. Here, we use results from carcass searches in the German federal state of Brandenburg to estimate the cumulative total number of Red Kite collisions with all wind farms in the study area. In order to assess their impact on the population we compare the results with mortality thresholds.

Methods

Study population

Red Kites breed throughout the 29,483 km^2 of the large German federal state of Brandenburg with an average density of 5.8 pairs/100 km^2 (Ryslavy et al. 2011). The best available estimate of spring population size is 1650–1900 breeding pairs (mean 1775; Ryslavy et al. 2011), thus representing 8% of the global population (Bird Life International 2012). Most birds migrate to south-west Europe in winter (Bird Life International 2012) but small numbers are found in Brandenburg in winter. During 1988–2010 numbers showed an annual decline of $-0.97 \pm 1.27\%$. We lack information about the non-breeding part of the population which might be considerable in a species recruiting at an age of 2–7 years (Newton et al. 1989). We considered two extreme cases: (i) a depleted population where low adult survival allows all sexually mature individuals to enter the breeding population; and, (ii) a stable population structure with non-breeders. We created a single sex Leslie matrix model (Caswell 2001) to estimate stable age distribution to calculate the size of the stable population, and growth rates. We used published information on survival (Table 1) and reproductive output (1.7 young per female) from Germany (Nachtigall 2008; Schönfeld 1984) to parameterise the model in programme ULM (Legendre & Clobert 1995). Data on recruitment age in Germany are scarce, so we chose values for age-specific breeding probability such that they approximated the only available small sample (Nachtigall 2008) as well as the findings of Newton et al. (1989).

Wind farm and collision data

Documented carcass searches were available from 69 wind farms including 617 turbines with rotor diameters of 40–100 m (mean 69 m) and nacelle heights of 41.5–160 m. Turbines were searched in all parts of the study area (Fig. 1). Between 2001 and 2011 a total of 7428 searches around 1–31 (mean 8.1) turbines per event were reported. Search intervals varied from 1 to 188 days with a median of two days (mean 5.3). Information on wind farms in Brandenburg (number, location, nacelle height and rotor diameter of turbines with a total height of 50 m or more) where taken from the database of the Landesamt für Umwelt, Gesundheit und Verbraucherschutz Brandenburg (LUGV).

Collision model

To estimate collision rate, i.e., the monthly number of Red Kite fatalities per wind turbine, we used a hierarchical model similar to the model used by Korner-Nievergelt et al. (2013). Our model consists of two sub-models, one for the collision process and one for the search process.

In the search process sub-model, we assumed the number of carcasses found per month and wind farm to be binomially distributed, $Y_i \sim \text{binomial}(N_i, p_i)$, with N_i equalling the number of Red Kites killed during month i and p_i being the probability that a Red Kite killed is later found by a searcher, henceforth called detection probability.

To estimate detection probability, we laid out a total of 20 Common Buzzards *Buteo buteo* as a surrogate for Red Kites at four of the wind farms studied. From repeated visits by different persons at intervals of 1–7 days until 15–30 days after placement, persistence probability s and searcher efficiency f were estimated with a Cormack–Jolly–Seber model in MARK (White & Burnham 1999). The estimates and 95% confidence intervals were transformed into Beta distributions, $\text{Beta}(a^s, b^s)$, and $\text{Beta}(a^f, b^f)$, respectively. We combined the average interval between the searches d, estimated carcass persistence probability s, and searcher efficiency f using the method of Korner-Nievergelt et al. (2011), to obtain the probability p_{100} that a Red Kite that was killed during a specific month was found by a searcher during that month given it was in the searched area. This method assumes that carcasses not found during one search can be found during another search with the same probability, given they persist.

Finally, p_{100} was multiplied by the proportion of carcasses expected to lie in the searched area, a. The proportion a was obtained from the search radius r^s and information about the proportion of area searched within the search radius. First, we obtained the proportion of carcasses within r^s based on the results from Hull and Muir (2010), who applied ballistics theory to model the distribution of carcasses around wind turbines. For large birds, they show a uniform distribution of carcasses from the tower up to a maximum distance of 85–95 m depending on turbine size. We fitted a standard linear regression of this maximum distance on rotor radius, as a proxy of turbine size, to obtain the maximum distance for each rotor radius in our data:

Maximum distance $= 69.90 + 0.45 \times$ rotor radius

A uniform distribution of carcasses between the tower and the maximum distance corresponds to a decreasing carcass density with the factor $1/(\pi r^2)$. We obtained the proportion of carcasses within the search radius r^s as the ratio of r^s and the maximum distance if r^s was lower than the maximum distance, or 1 otherwise. The ballistic model predicts a limited proportion of carcasses beyond the maximum distance, and injured birds may move even further away from the turbine. We therefore performed our estimation under two different scenarios: (i) all victims are found within the maximum distance; (ii) 15% of the victims die outside this area.

Second, because most turbines were situated in farmland, large parts of the area were unavailable for carcass searches from spring until harvest. For those searches j where this was not documented, we assumed a uniform distribution between the season specific minimum ($a^{s\text{-min}}$) and maximum ($a^{s\text{-max}}$) possible proportions of the area searched within r^s (April 0.1–1, May–July 0.05–0.8, August–March 0.8–1). These values were multiplied with the proportion of carcasses within the search radius to obtain the proportion of carcasses in the searched area a_j.

To take into account uncertainty in f, s, and a we performed a Monte Carlo simulation. We first drew 1000 random values
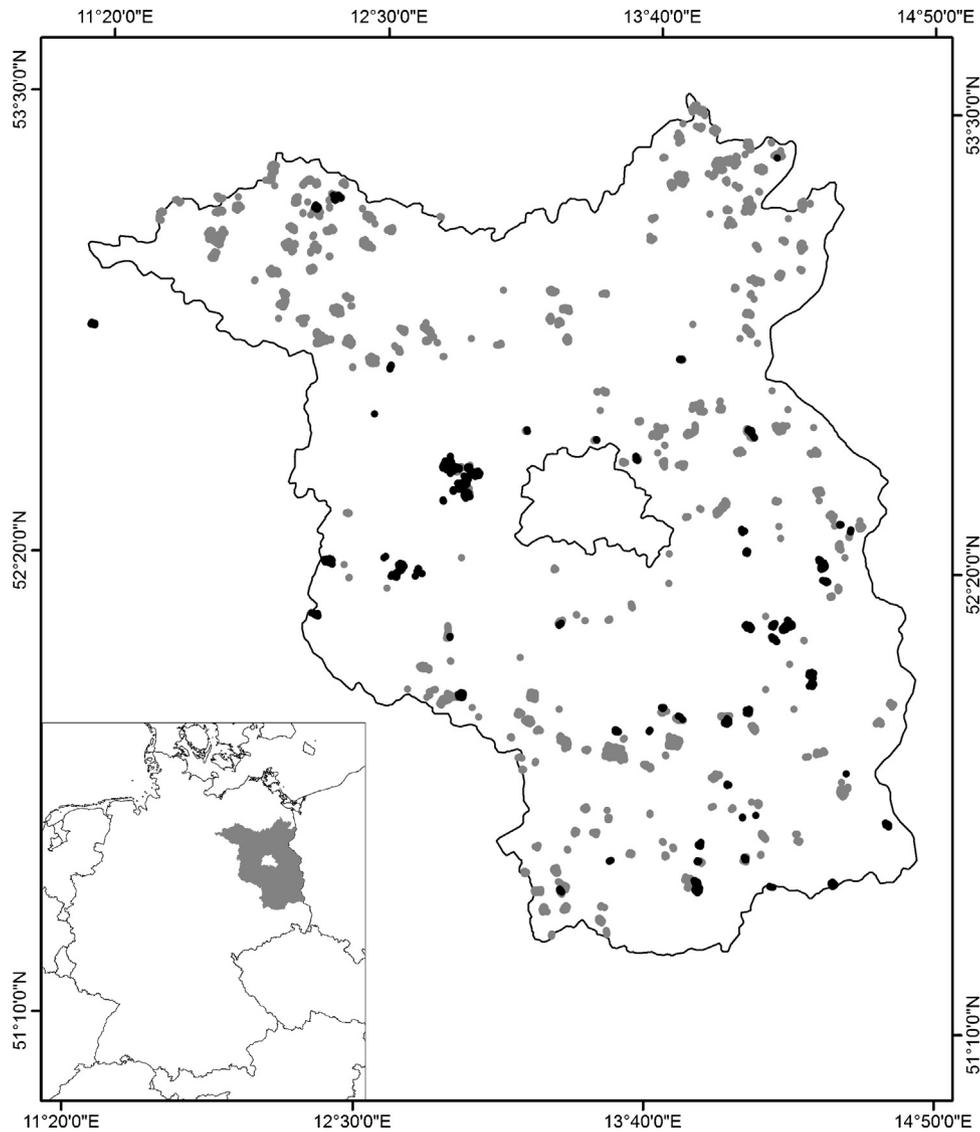
Fig. 1. Locations of turbines operating in 2012 used for extrapolation, turbines searched for carcasses shown in black. The inset shows the location of the study area (shaded) in Germany.

from Unif(a^{s_min}, a^{s_max}) for each search and transformed the monthly averages to a_i^*. Then, we drew 1000 random values s_i^* and f_i^* from Beta(a^s, b^s), and Beta(a^f, b^f). Based on these random values, we obtained 1000 possible values for detection probability p_i. We transformed the distribution of the 1000 p_i^* values into Beta distributions, and then used the parameters of these distributions (a_i^p, b_i^p) as prior information for detection probability.

The collision process sub-model predicted the Red Kite collision rate for each wind farm from rotor diameter, Red Kite breeding density and month. The number of collisions per wind farm and month was assumed to be Poisson-distributed with $\lambda_i n\text{WEA}_i/10$ as expected value: $N_i \sim \text{Poisson}(\lambda_i n\text{WEA}_i/10)$, where $n\text{WEA}_i$ was the number of turbines searched and λ_i the expected number of collisions at 10 wind turbines per month. We chose the unit of 10 wind turbines to avoid λ_i values close to the lower bound of the parameter space at zero, which could produce problems in the model fit. The logarithm of λ_i was modelled linearly dependent on Red Kite breeding density D_i, on season (a factor with the nine levels winter (November–February), and all remaining months, April was used as the reference level), and on rotor diameter (linear and quadratic

effect). We further included a normally distributed random wind farm effect $wf_j \sim \text{Norm}(0, \sigma_{wf})$ and an additional variance parameter $\varepsilon_i \sim \text{Norm}(0, \sigma_\varepsilon)$ to allow for potential overdispersion.

$$\log(\lambda_i) = \beta_0 + \beta_1 D_i\,\text{summer}_i + \beta_2\text{winter}_i + \beta_3\text{march}_i + \beta_4\text{may}_i$$
$$+ \beta_5\text{june}_i + \beta_6\text{july}_i + \beta_7\text{august}_i + \beta_8\text{september}_i$$
$$+ \beta_9\text{october}_i + \beta_{10}\text{rotor}_i + \beta_{11}\text{rotor}_i^2 + wf_j + \varepsilon_i$$

We allowed the breeding density D to affect the number of collisions only during the summer months, by multiplying this predictor by an indicator variable, *summer*, which included the months March–September. All numeric covariates were standardised to mean 0.0 and standard deviation 1.0. We used diffuse prior distributions, i.e., Uniform(−10, 10) for the β coefficients of the linear predictor, and Gamma(0.01, 0.01) for the variance parameters.

The model was fitted using Markov chain Monte Carlo simulations. Two Markov chains were run for 11,000 iterations and burn in was set to 6000, thus 10,000 random values could be used to

Table 2
Parameter estimates of the collision probability model. Means and 95% credible intervals are shown for the two scenarios of carcass distribution described in the text. Significant parameters are indicated by credible intervals not including zero and shown in bold.

Parameter	All victims within the area assumed by the ballistic model		15% of the victims outside	
	Estimate	95% CrI	Estimate	95% CrI
Intercept (β_0)	**−1.34**	**−2.77** to **−0.23**	**−1.12**	**−2.49** to **−0.07**
Breeding density (β_1)	0.21	−0.24 to 0.64	0.23	−0.19 to 0.63
Winter (β_2)	**−3.88**	**−7.00** to **−1.69**	**−3.88**	**−7.02** to **−1.74**
March (β_3)	**−2.56**	**−5.70** to **−0.37**	**−2.57**	**−5.84** to **−0.43**
May (β_4)	−0.06	−1.33 to 1.26	−0.07	−1.32 to 1.11
June (β_5)	−0.82	−2.45 to 0.64	−0.82	−2.43 to 0.63
July (β_6)	−0.75	−2.18 to 0.64	−0.69	−2.10 to 0.63
August (β_7)	**−1.62**	**−3.08** to **−0.26**	**−1.61**	**−3.04** to **−0.29**
September (β_8)	**−2.53**	**−4.57** to **−0.85**	**−2.50**	**−4.50** to **−0.84**
October (β_9)	**−6.51**	**−9.84** to **−2.49**	**−6.51**	**−9.83** to **−2.41**
Rotor (β_{10})	−0.59	−1.32 to 0.05	−0.58	−1.29 to 0.02
Rotor^2 (β_{11})	−0.63	−1.44 to 0.1	−0.61	−1.42 to 0.09
Between-wind farm standard deviation (σ_{wf})	13.45	0.5 to 64.93	21.25	0.35 to 82.71
Overdispersion (σ_ε)	9.41	0.27 to 64.98	22.51	0.35 to 153.7

describe the joint posterior distribution of the model parameters. Convergence was assessed by the Brooks–Gelman–Rubin statistics (Brooks and Gelman, 1998).

The estimate for the total yearly number of fatalities was obtained from the model formula

$$\lambda_i = \exp(\beta_0 + \beta_1 D_i\, \text{summer}_i + \beta_2 \text{winter}_i + \beta_3 \text{march}_i + \beta_4 \text{may}_i$$

$$+ \beta_5 \text{june}_i + \beta_6 \text{july}_i + \beta_7 \text{august}_i + \beta_8 \text{september}_i$$

$$+ \beta_9 \text{october}_i + \beta_{10} \text{rotor}_i + \beta_{11} \text{rotor}_i^2)$$

We calculated the expected number of fatalities for each month for each wind farm in Brandenburg, i.e., a group of n wind turbines with equal rotor diameter and equal Red Kite breeding density, by the model formula. The resulting twelve λ_i values were multiplied by $n/10$ to adjust the value for the number of wind turbines and summed up to a yearly total. Finally, we summed up the estimates for all wind farms to obtain an estimate of the total number of fatalities. To describe the uncertainty of this estimate, we repeated the procedure for each of the 10,000 sets of parameter values and used the 2.5% and 97.5% quantiles of the 10,000 different estimates for the yearly total as a 95% credible interval. The credible interval expresses our uncertainty about the average number of fatalities.

We used R 2.15.0 for data handling and standard analyses. The model was fitted in OpenBUGS (Spiegelhalter et al. 2007) using the R-interface BRugs (Thomas et al. 2006).

Potential biological removal

We calculated mortality thresholds using the potential biological removal (PBR) approach (Wade 1998). Mortality limits are calculated as

$$\text{ML} = N_{\min} \frac{1}{2}(\lambda_{\max} - 1)f$$

where $N_{\min}$ is a minimum estimate of population size, $\lambda_{\max}$ the maximum possible growth rate of the population when no density dependence is acting, and the recovery factor f a value between 0.1 and 1.0. By using $N_{\min}$ and f, the equation acknowledges uncertainty or potential bias in the estimates of population size and growth rates (Wade 1998). We used the lower bound of the regional population estimate as $N_{\min}$. The maximum growth rate $\lambda_{\max}$ for the study population was estimated in a matrix model assuming unlimited recruitment at age 2 and a breeding productivity of 2.3 young per female which equals brood size at ringing age (Nachtigall 2008, own unpublished results).

We applied the modelling procedures from Wade (1998) to select appropriate values of f for populations with $\lambda_{\max} = 1.4$ with a coefficient of variation for the sampling distribution of $N_{\min}$ of 0.4. According to the capacity goal after Wade (1998) we ran simulations with 2000 replicates for 100 years. In addition to $\theta = 1$ (Wade 1998) we used $\theta = 1.5$ according to the estimate for Red Kites by Eberhardt and Breiwick (2012).

Results

Population size

A depleted population without mature non-breeders would consist of 7841–9029 (mean 8435) individuals. A matrix model showed that under a stable age distribution breeding adults would represent 36% of the total population after the breeding season (Table 1). The post-breeding population size could then be estimated as

Total number of individuals = number of breeding pairs × 2/0.36 resulting in 9270–10,674 (mean 9972) individuals.

Collision model

A total of 34 fatalities were found during 7410 documented carcass searches, i.e., carcasses were recorded in 0.46% of the searches. Six carcasses were older than 40 days and were excluded from the estimation of monthly fatality rates. Among opportunistic fatality records not restricted to a certain search radius, nine out of 49 victims were found 150–1000 m from the next turbine. Thus, at least 18% of the colliding birds were found outside of the area expected in the model.

Our experiments with Buzzard carcasses resulted in estimates of $s = 0.976$ (95% confidence interval 0.950–0.988) for daily persistence probability, corresponding to an average persistence time of 41 days, and $f = 0.508$ (0.287–0.725) for searcher efficiency.

The estimated probabilities that a Red Kite is found after colliding ranged between 0.01 and 0.85 (mean = 0.35, sd = 0.17) if no birds moved away from the turbine, and from 0.01 to 0.72 (mean = 0.30, sd = 0.14) if 15% of the birds died beyond the maximum distance. Using these estimates in the collision model, we obtained positive (though not significant) correlations of collision risk with Red Kite breeding density and with rotor radius (Table 2). Negative parameter estimates in Table 2 indicate that the collision risk was significantly lower from August until March, while it was highest in April (the reference month) and May.

In the more realistic scenario, where 15% are found beyond the maximum distance, a total number of 308 Red Kites was estimated

Table 3
Estimated annual numbers of Red Kite fatalities for different stages of wind farm development and population sizes (mean number of individuals, see text).

Scenario	Mean	95% Credible interval	Proportion of population (%)	
			Depleted	Stable
Turbines operating 2012 (3044)				
All victims within the area assumed by the ballistic model	241	88–407	2.9	2.4
15% outside	308	159–488	3.7	3.1
Turbines operating or approved (3294)				
All victims within the area	251	91–424	3.0	2.5
15% outside	320	165–508	3.8	3.2
All turbines (3749)				
All victims within the area	258	94–439	3.1	2.6
15% outside	330	168–530	3.9	3.3

Table 4
Mortality limits after the potential biological removal method for different population sizes (see text) and estimated maximum growth rates λ_{max}.

	λ_{max}	Number of individuals	Proportion of population (%)
Depleted population			
E Germany	1.29[a]	339	4.0
Great Britain	1.39[b]	455	5.5
Stable population			
E Germany	1.29	400	4.0
Great Britain	1.39	538	5.4

[a] Matrix model (see text).
[b] Evans et al. (1999).

to collide annually at the 3044 turbines operating by the end of 2012, representing 3.1% of all individuals in the post-breeding population under a stable age distribution (Table 3).
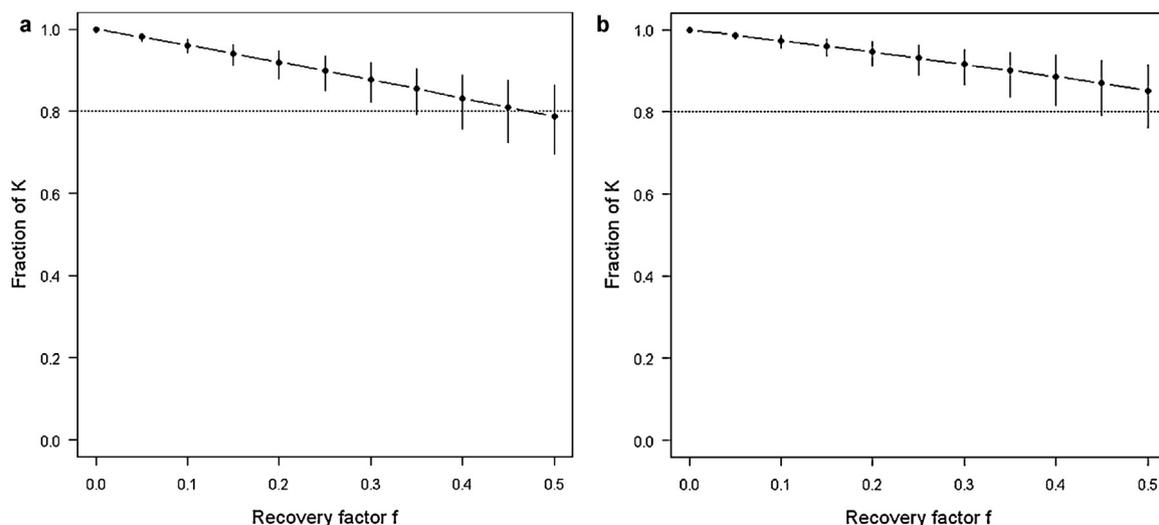
PBR thresholds

The estimated maximum possible growth rate λ_{max} of the study population was 1.29. We also calculated PBR with the highest published growth rate for British Red Kites (Table 4). Simulations showed that using f values greater than 0.3 strongly increased the risk that the population would drop below 0.9 K and even below 0.8 K (Fig. 2). Using $f = 0.3$ we obtained mortality thresholds of 4.0% in the East German population and 5.5% for a thriving population without additional losses (Table 4).

Discussion

Red Kite mortality

Like various, but not all, other raptor species, breeding Red Kites are not displaced from their foraging habitats by wind turbines (Madders & Whitfield 2006). In such species most case studies did not reveal effects of wind turbines on local breeding numbers (Stewart et al. 2007; Pearce-Higgins et al. 2012). In an extreme case, however, Griffon Vultures *Gyps fulvus* were forced to forage close to wind farms and temporarily experienced a 30% decrease in adult survival and a population decline (Martínez-Abrain et al. 2012). Collision models predict that fatalities at large numbers of wind turbines may negatively affect the long-term viability of Red Kite populations, too (Schaub 2012). Our estimates based on actual carcass records confirm for the first time that cumulative wind turbine fatalities at existing wind farms may reach such a level.

Altogether 10% of 101 Red Kite fatalities of known age were older than one year (own unpublished data), hence collision mortality has a clear potential to influence population growth, which is most sensitive to changes in adult and subadult survival (e.g., Whitfield et al. 2004). Red Kite populations have long been subject to various types of human induced mortality (Knott et al. 2009; Langgemach et al. 2010; Smart et al. 2010). This has led to extinction in England and Scotland (Evans et al. 1999) and still limits the growth of regional populations (Smart et al. 2010). In long-lived raptors, an additional adult mortality of 3–5% can already inhibit population growth (Whitfield et al. 2004) or increase extinction

Fig. 2. Simulated population size (mean with 90% confidence interval) after 100 years in PBR simulations after Wade (1998) for a population with $\lambda = 1.4$ and different values of f, assuming (a) $\theta = 1.0$ and (b) $\theta = 1.5$.

risk (Carrete et al. 2009). The thresholds calculated with the PBR method, used as a rapid assessment tool here, indicate that this is probably also true for Red Kites. Since wind energy production is rapidly expanding in all European countries (Tellería 2009; Pullen & Sawyer 2011) it will become a potential threat throughout the species' breeding and wintering range in the near future.

Fatality estimation

Detection probability primarily determines the reliability of fatality estimates (Korner-Nievergelt et al. 2011). Both persistence probability s and searcher efficiency f increase with carcass size (e.g., Smallwood 2007; Grünkorn et al. 2009). Standard methods exist for carcass removal trials, but sample sizes for large raptors are limited by the availability of carcasses. At the time of our fieldwork it was common practice to use estimates from previous studies (e.g., Smallwood & Thelander 2008). Our estimates of s and f are close to, or slightly higher than, previous estimates for similar sized birds in Germany (Grünkorn et al. 2009; T. Grünkorn and J. v. Rönn pers. comm.). Search intervals were on average much shorter than carcass persistence, hence carcasses were available for detection in several consecutive searches. We treated s and f as constant because of a lack of detailed knowledge, while they may actually change over time (Bispo et al. 2013; Huso 2010). For short search intervals our approach could have led to an underestimation of the fatality rates (Korner-Nievergelt et al. 2011). The largest uncertainty was caused by cases for which the areas within the search radius covered during each search were not reported. We recommend more carcass removal trials and regular reporting of areas searched to improve the precision of fatality estimates in future studies.

Conservative assumptions were also applied in using the ballistic model from Hull and Muir (2010) to predict the spatial distribution of carcasses. We did not apply a correction for the fact that towers in Brandenburg are generally higher than in Hull and Muir (2010) which should result in a higher maximum distance, and we ignored the distant area where <10% of the carcasses are expected according to their model. Hull and Muir (2010) also found carcasses of large birds beyond the predicted area, mainly because some injured birds moved away from the turbine before dying. While during the searches carcasses were only found within the search radius, opportunistic fatality records up to 1000 m away from the turbine (see above) suggest that even more than 15% of the collided Red Kites died outside this radius.

Population size

Sound knowledge on population size is limited to the breeding part of the population. The proportion of mature non-breeders will depend on population status, with more non-breeders in stable populations with density-dependent regulation. Based on the scarce information on age of recruitment in Red Kite populations not strongly increasing, we created two scenarios to represent the extreme ends of the possible population structure. No evidence exists that the population in Brandenburg is severely depleted, and we assume the stable population case to be more realistic.

Potential biological removal

Population models are needed to more precisely estimate the maximum sustainable level of additional mortality, while other threats in the breeding and wintering range persist. At present, such a model for German Red Kites would require several arbitrary assumptions particularly on density dependent regulation, while PBR does not (Dillingham & Fletcher 2008). The choice of f is critical for the outcomes of PBR calculations. Available recommendations are based on simulations for marine mammals with $\lambda_{max} = 1.12$ or

less (Wade 1998). Our simulations lend some support to the recommendation of $f = 0.3$ for near threatened populations (Dillingham & Fletcher 2008). The risk for a population not to reach 0.9 K, however, was already substantial for $f = 0.3$, indicating a need for further simulations to guide the choice of f for populations with $\lambda_{max} > 1.12$.

A second critical choice concerns the population growth rate. Using parameters suitable for East German Red Kite populations resulted in a lower growth rate than found for Red Kites in a thriving British breeding population (Evans et al. 1999). Most Red Kites from Central Europe migrate to France and Spain for wintering, where they may suffer from additional persecution or collision losses (Langgemach et al. 2010), whereas British birds usually winter in the breeding areas. Even there, only populations which are not subject to illegal killing reach growth rates of 1.39 (Smart et al. 2010). While a λ_{max} of 1.39 probably represents the maximum growth rate under optimal conditions, the resulting PBR threshold would have to be compared to the yet unknown total additional mortality, rather than only to wind farm related fatalities in the breeding area. For comparison with our fatality estimates the appropriate PBR thresholds are those for the Eastern German population with $\lambda_{max} = 1.29$. These thresholds were inside the 95% credible interval of our fatality estimates and may be exceeded by the numbers of fatalities when further turbines are constructed.

The Red Kites in Brandenburg are part of a contiguous Central European breeding population with a regular exchange of individuals with adjacent areas (Nachtigall 2008). Applying the PBR concept which assumes a closed population is still justified for two reasons. Firstly, surrounding Red Kite populations show similar densities and trends, thus not suggesting a net immigration or emigration of birds to or from Brandenburg. Secondly, they are undergoing the same development of wind energy production, and equally high fatality rates may be expected in these areas today or in future.

Implications

The Red Kite is listed on Annex I of the EU Wild Birds Directive (EEC/79/409) which obliges member states to achieve or maintain the species in a favourable conservation status. To prevent effects from cumulative fatalities at multiple wind farms on conservation status, an assessment at the national or global population level, and legislation implementing upper thresholds of acceptable mortality would be required. Further wind farm development in Brandenburg and surrounding areas should therefore be accompanied by (i) comprehensive population modelling to assess the sustainable level of fatalities, (ii) assessments of state-wide cumulative impacts, and (iii) effective mitigation. Mitigation may include limits to the construction of new turbines (Schaub 2012) and creation of Red Kite foraging habitat at a safe distance from existing wind farms (Martínez-Abrain et al. 2012).

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