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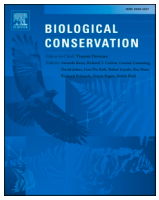
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Reduced survival in a soaring bird breeding in wind turbine proximity along the northern Baltic Sea coast

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ABSTRACT

Wind power plays a vital role in global climate action and plans for new turbines in the Baltic Sea region are underway. However, it is crucial not to overlook the environmental impact on wildlife, which can be difficult to quantify, especially in elusive and hard-to-track species. One of these species is the white-tailed eagle, a conservation success story that faces a significant collision risk with rotor blades, particularly as turbines are being constructed in its coastal habitats. To assess the effect of wind turbines on survival of territorial adults, we genotyped DNA from adult feathers collected at nests between 2010 and 2022. By tracking individuals across years, we measured survival and breeding dispersal in relation to wind turbine presence. Turbines within a 5 km radius of nests were found to reduce annual survival rates by 7.6 %, while resighting probability and breeding dispersal probability were unaffected. The proportion of territories exposed to wind turbines is currently low (4–5 %), mainly because recent construction sites have been further inland. However, future projections suggest an increase in territory exposure, indicating potential for population-level risks. This highlights the need for substantial safety buffers around nest sites (preferably >5 km) to protect the breeding white-tailed eagle population. Currently, a comprehensive understanding of the large-scale impact on the Baltic Sea white-tailed eagle population is lacking. As wind power expands, it is necessary to consider its impact on wildlife and we recommend conducting ongoing environmental assessments to monitor and adapt conservation measures.

1. Introduction

To meet global climate change goals, the production of renewable energy is increasingly expanding to satisfy energy demands (Bouckaert et al., 2021; Global Wind Energy Council, 2022). Wind turbines are a popular option to produce electricity and their construction is encouraged in many countries (Global Wind Energy Council, 2022). They can however have detrimental effects on wildlife (Loss et al., 2013; Wang and Wang, 2015): During construction, the surrounding area can be subject to disturbance due to increased human presence (Pearce-Higgins et al., 2012). Animals might avoid wind turbines also after construction (Colman et al., 2012), which can lead to barrier effects during migration (Cabrera-Cruz and Villegas-Patraca, 2016; May, 2015) or loss of habitat (Campedelli et al., 2014; Coppes et al., 2020). A threat to flying animals is direct mortality due to collision (Gaultier et al., 2020; Loss et al., 2013) and several mitigation strategies have been developed to reduce collision risk (reviewed by May et al., 2015; Sayed et al., 2021) but it is unclear how widely they are used or how effective they are.

Although mortalities due to wind turbines are estimated to be lower than those associated with fossil fuel energy (Sovacool, 2009), a concern is that they disproportionately affect long-lived and slowly reproducing species. Species such as vultures, eagles and other raptors are often few in numbers and already of conservation concern (Watson et al., 2018). As a result, their populations are hit hard by the loss of even a few individuals (Tack et al., 2017). Mortality rates can be considerable, i.e. wind collisions were found to be responsible for a reduction of survival of up to 30 % in cinereous vultures *Aegypius monachus* (Martínez-Abraín et al., 2012). Many more species are known to be at threat to collide with wind turbines, but detailed insight into the impact on their survival is currently lacking.

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Although the threat of wind turbines on some bird species groups is widely acknowledged (i.e. Diffendorfer et al., 2021), it is often difficult to quantify their effect on survival, with most studies focussing on observed mortality events. This is due to vulnerable species often being shy and elusive, and prone to human disturbances. Furthermore, they often have large home ranges and thus breed at low densities. Due to low detection rates, it can be a challenge to obtain data in meaningful quantities (Newton et al., 2016; Watson et al., 2018). Here, non-invasive genotyping methods can help to monitor populations (e.g. Kylmänen et al., 2023; Rudnick et al., 2009; Penttinen et al., 2024), and allow to quantify how wind power might affect breeding populations.

The white-tailed eagle *Haliaeetus albicilla* is a success story of conservation in Northern Europe. In Finland, after being close to extinction in the 1970s due to a long history of persecution and thereafter environmental contaminants, the population recovered in the 1990s and 2000s (Högmander et al., 2020; Saurola et al., 2003; Stjernberg et al., 2005). Today, the species is no longer endangered in Finland (Högmander et al., 2020), but threats remain of e.g. bioaccumulating toxins (Ekblad et al., 2021; Isomursu et al., 2018) and wind power (Balotari-Chiebao et al., 2016a, 2016b).

In Finland, wind power will be a main source of energy in the future. White-tailed eagles often soar using thermal uplifts at wind turbine height on land (Tikkanen et al., 2018) and experience high collision risks due to their incomplete avoidance behaviour towards wind turbines (Dahl et al., 2013). Prime foraging and breeding habitats for white-tailed eagles are also ideal wind turbine locations (Tikkanen et al., 2018), thus the conflict of wind power and white-tailed eagle conservation needs to be more thoroughly explored. Wind turbine proximity is associated with lower breeding success in white-tailed eagles (Balotari-Chiebao et al., 2016b), but it is not known whether this is due to higher adult mortality or avoidance behaviour leading to nest abandonment. Studies at the Norwegian Smøla wind park have shown that white-tailed eagles are negatively affected by wind turbines near their nests, leading to lower breeding success and nest abandonments (Dahl et al., 2012). This area is, however, only one wind farm in a locally dense white-tailed eagle population and the results may not be the same across different environments. We thus lack a complete understanding of how wind turbines affect breeding white-tailed eagles on larger spatial and demographic scales. This knowledge gap is important to fill to make more informed conservation decisions.

In this study, we use the white-tailed eagles breeding in the Baltic Sea coast as a study system, for which unique data is available to quantify the impact of wind turbines on survival and displacement of a large soaring raptor. We use non-invasive genotyping to identify adults at nests in subsequent years and supplement this information with visual identification of individually ring-marked eagles. As apparent survival estimated from mark-recapture data is the combined measure of mortality and emigration (Newton et al., 2016), we also quantify breeding dispersal probability in subsequent years and breeding dispersal distance between alternative nests in relation to wind turbine presence. Raptors might abandon their territories (May et al., 2013) or use alternative nest sites within their territory in response to disturbance or nest failure (Ontiveros et al., 2008; Postupalsky, 1974). Therefore, this might represent a measure of potential behavioural responses like avoidance or displacement which are caused by wind turbines. With this combined approach, we aim to better understand the impact of wind turbine construction on breeding individuals of a large, soaring raptor on a large spatial scale, and to identify appropriate safety distances between wind turbines and nests. Our results will help make more informed construction decisions in the face of the most recent and future advances in wind turbine construction.

2. Material and methods

2.1. Field data and adult feather collection procedure

The monitoring of the Finnish Baltic white-tailed eagle population has been conducted by the White-tailed Eagle working group that has operated under World Wildlife Fund (WWF) Finland (1973 – 2019) and under the Osprey-foundation (2020 onwards). Nests were visited annually during the breeding season, where nesting status ('outcome') was determined from the ground, by using drones or by climbing the nesting tree. Nest status was determined as 'inactive' when there were no fresh signs of nesting activity (nest was not built upon with fresh sticks), and 'active' when there were eggs or nestlings present, or there were fresh signs of activity, i.e. in minimum the nest had fresh sticks and branches brought in. A 'successful attempt' contained nestlings of at least ringing age (3 weeks or older). Every year, all known territories (453 in 2010; 835 in 2022) were visited by volunteers. Nests were visited in sequence and if the active nest with nestlings was found, the other known alternative nests within a territory were not visited. During nest visits, nestlings were ringed and a feather sample (2–3) was collected from them (since 2003), and adult feathers were collected in the nest or beneath it. About 68 % of all feathers collected were from females, therefore female white-tailed eagles have higher encounter rates based on genotyped feathers. Adult feather collection started in 2001 (with variation between regions), but only from 2010 onwards, the number of feathers (n = 2441 different year-territory combinations) across the entire study area were considered sufficient for an analysis. Over the entire study period (observation years 2010–2022), 586 different nests were sampled in coastal areas in Finland (Fig. 1). Feathers were stored in envelopes before DNA extraction (see Supplementary

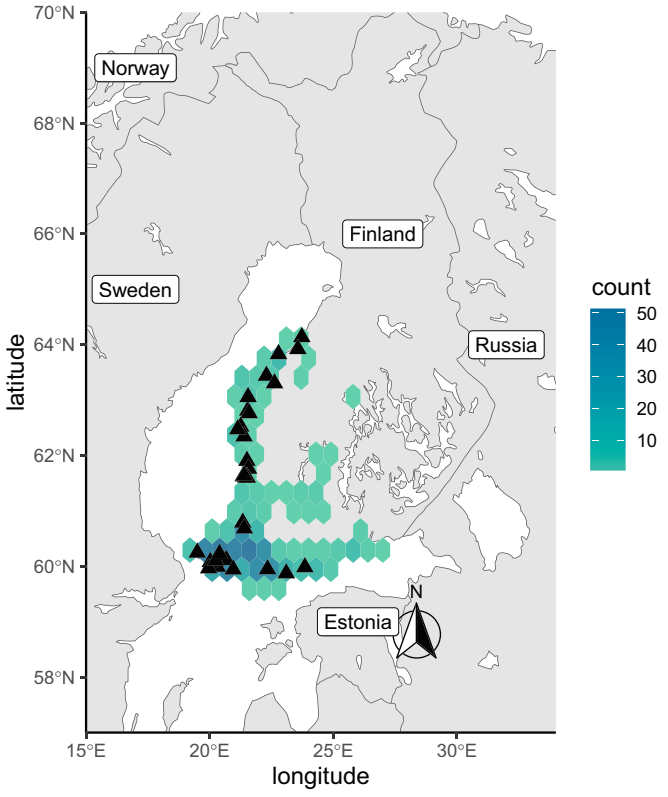


Fig. 1. Map of the Baltic Sea area showing the density of white-tailed eagle nests sampled in this study in coastal areas of Finland (light green: low density, dark blue: high density). Black triangles are wind turbines that have a nest in a 5 km radius around the nest site. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

material for extraction and genotyping protocols). Genetic identification was supplemented with visual resightings at nests ($n = 53$ individuals, of which 15 were confirmed by genotype). Visual identification was based on high-resolution photographs and individuals were identified based on the engraved colour rings that they had obtained as nestlings. White-tailed eagle nestlings were also genotyped from feather DNA from a subsample collected in 2008–2014 and in 95 % of all cases, the adult feathers collected at the nest were from a parent bird ($n = 155$, detailed analysis not shown). This also corresponds to observations from other study areas that show that the majority of feathers collected belong to the nest owner (Rymešová et al., 2020).

2.2. Wind turbine presence around nests and territories

Wind turbine longitudes and latitudes were obtained from the Finnish National Land Cover database (https://www.maanmittauslaitos.fi/, accessed 06/2023) and information on construction and demolition years was obtained from the Finnish Wind Power Association (https://tuulivoimayhdistys.fi/en/, accessed 06/2023). The first turbines were built in 1991, but a construction boom happened in the mid-2010s and intensified at the beginning of 2020 (Fig. S1). Currently, 6 GW are produced by onshore wind turbines, and Finland plans to increase onshore production to 13–36 GW by 2035 and up to 76 GW by 2045 (FINGRID, 2022). The planned projects represent 21–63 GW (Finnish Wind Power Association, 2023). Locations of wind turbines were used to obtain their presence in different radii around nests and territories. Since 2009, white-tailed eagle nest locations have been considered during environmental impact assessments (EIAs), and turbines are usually not built closer than 2 km from an eagle's nest. Thus, the sample was too small for 2 km buffers to be accounted for and for buffers of 3–4 km they were also quite small (<50 individuals; Tables S1, S2).

2.3. Statistical analysis of adult survival and breeding dispersal

We assessed consequences of wind turbine presence by analysing adult survival and potential displacement via breeding dispersal behaviour. We used seven different buffer radii to identify, at which distances wind turbine presence affected white-tailed eagles: 3 km, 4 km, 5 km, 6 km, 7 km, 8 km and 9 km radius around the nest site. It was necessary to examine the relationships at different radii, as no single radius was in advance known to be the most important. There can be quite considerable variation in the home range sizes for white-tailed eagles, and they can roam far from their nests when foraging (Krone et al., 2013). The seven buffers were fitted in contrasting models and compared to each other and null models (that did not contain any wind turbine-related variables) by second order Akaike Information Criterion (AICc). AICc is used to account for small sample sizes and becomes identical to AIC when large samples are present. A ΔAICc >2 was considered a threshold. If models differed by a ΔAICc >2, we performed a likelihood-ratio-test (LRT) to test whether the top model containing a wind turbine-related variable and the nested top null model statistically significantly differed from each other. A full model comparison list can be found in Tables S3–S5.

To evaluate how presence of wind turbines (presence or absence) in the vicinity of an individual's nest location affected annual apparent survival, we performed a Cormack-Jolly-Seber (CJS) survival analysis by using the 'marked' package v.1.2.6 (Laake et al., 2013). The seven alternative buffer variables around wind turbines were time-varying and accounted for construction and demolition of wind turbines over time. We assumed that wind turbines were present if construction had finished in the observation year.

The capture history for the CJS analysis is a sequence of '0' (unobserved state) and '1' (observed state, when an individual is documented alive). An individual was considered to be alive if it was identified visually or based on its genotype in an observation year. To determine whether a bird was nesting near wind turbines, it was necessary to decide the nesting location of each individual in each observation year and for each state: while the nest location of an eagle was known in years in which they were identified and considered to be alive (observed state) based on the location of its collected feather or visual resighting, their possible nesting location was unknown in years when an individual was not observed. In that case, assumptions had to be made about its nesting location: In the unobserved state, eagles can be alive and present at the same nest (but go undetected), be alive but have moved to an alternative (sometimes unknown) location or have died. This was resolved in the following way: In the unobserved state, we assumed that individuals were present at the same location since their last sighting and were failed to be identified. This is a reasonable assumption, as white-tailed eagles have overall high site fidelity and our study design made it likely that one partner of a pair was not identified. In case of observed movement to a new nest location, we assumed that eagles departed in the year after their last observation and had been nesting at the new nest location. These assumptions had to be made as the 'marked' package or alternatives do not allow any missing data. We considered the following fixed effects and their combinations: spatial location (latitude and longitude) and sex.

Resighting probability was modelled as 'sex' (factor, 'male' and 'female') as females are more likely to be genotyped from feathers found at nest sites, time (factor, 2010–2022), number of years a nest had been visited by volunteers during the observation period (mean = 8.7, numeric), nesting outcome ('inactive', 'active', 'successful' and 'unknown', factor; which will strongly influence the time parents spend around the nest site), whether an individual was identified visually or by genotype (factor, individuals that were identified by both methods were treated as visually identified, $n = 15$) and spatial location (mean latitude and longitude, as resighting effort might differ between volunteer groups). We used the package 'R2ucare' v.1.0.2 (Gimenez et al., 2017) to assess goodness-of-fit and 'RMark' v.3.0.0 (Laake, 2013) to export the data and estimate the overdispersion parameter median ĉ in MARK (White and Burnham, 1999).

To assess breeding dispersal behaviour, we used a two-fold approach, in which we first assessed the probability of using alternative nest sites in a generalized linear mixed model (GLMM, binomial family) and then distance between alternative nest sites in a linear mixed model (LMM) that assesses how far white-tailed eagles move in case of breeding dispersal events. While the GLMM allowed us to assess whether eagles were more likely to abandon their nest site, the LMM gave insight into whether eagles were responding to wind turbines by using alternative nest sites within their territories or abandoning their territories altogether. In the GLMM comparison, the response variable was breeding dispersal ('0' = eagle remained at same nest, '1' = eagle moved to another nest). Fixed effects considered were sex and spatial information (nest latitude and longitude, both numeric). In the LMM, the response variable was dispersal distance (log-transformed). We considered covariates sex, spatial information, the number of years between observations ('gap'). Random terms in all LMMs and GLMMs were individual identity and year.

For all analyses, we used R v.4.3.0 (R Core Team, 2023). For data organization and visualization, we used the 'tidyverse' package v.2.0.0 (Wickham et al., 2019); distances between nests and wind turbines were calculated using the 'geosphere' package v.1.5–18 (Hijmans, 2021), the map was created with 'sf' v.1.0–13 (Pebesma, 2018), 'rnaturalearth' v.0.3.2 (South, 2017a), 'rnaturalearthdata' v.0.1.0 (South, 2017b), and 'ggspatial' v.1.1.8 (Lüdecke, 2018). We used the 'lme4' package v.1.1–33 to fit mixed models (Bates et al., 2015) and the 'DHARMa' package v.0.4.6 (Hartig, 2022) to validate model fit.

2.4. Estimating population-level mortality events

Lastly, we assessed what wind turbines could mean in terms of adult breeder mortality events on the population-level. To obtain estimates for

the period 1991–2022, we used the population data of white-tailed eagles inhabiting coastal habitats (excluding territories from Lapland) and assessed how many territories had a wind turbine within the 5 km critical buffer (location was quantified as the mean position of all known nests of the territory). To calculate expected number of mortality events on the population-level, we used the survival estimates and standard errors obtained from the CJS analysis. To obtain an approximate range of minimum and maximum mortality events, estimation was done first for all known territories (active and inactive, 'maximum mortality events') and active territories only ('minimum mortality events'). We assumed that both male and female of a pair experienced equal probability of mortality after we first explored the interactive effect of wind turbine presence and sex in our CJS models, and found no effect. To obtain an estimate for mortality events caused by wind turbines, we first calculated how many mortality events could be expected in the observed population under wind turbine exposure in 5 km radius. Then, we considered a scenario without any wind turbines ('no-wind-turbine-scenario') and estimated the mortality events that would naturally occur in their absence. By subtracting the estimated mortality events in the no-wind-turbine scenario from the actual estimated mortality events, we obtained the estimated number of mortality events specifically caused by wind turbines. Simulation was done using these known parameters (number of pairs based on number of territories and survival rates) with 1,000,000 iterations. Percentual changes were calculated with the equation:

$$\text{change}\ (\%) = \frac{(a-b)}{a} * 100$$

with 'a' being the larger value and 'b' the smaller value. When calculating the change in apparent survival due to wind turbine presence, the survival estimate without a wind turbine is 'a' and the survival estimate in wind turbine presence 'b'. When estimating the population-level morality events, the estimated mortality events in the real population are 'a' and the mortality events in the no-wind-turbine scenario are 'b'.

2.5. Future wind power exposure

For future predictions of how many of the current territories may be affected by the wind turbines that will be built by 2030 (Fig. S2), we used the location and number of planned turbines of wind farms (that contain multiple wind turbines) obtained through the Finnish Wind Power Association (2023). Currently, there are 401 onshore and offshore (near coastal areas) wind farms known to be in construction or planning stages in addition to the existing 1468 (Finnish Wind Power Association, 2023). We assumed that the planned wind turbines would be built around their approximate construction site in a square grid, with individual turbines 900 m apart from each other. If the number of wind turbines within a wind farm was unknown ($n = 9$), the number was assumed to be 21, which is the average size of planned wind farms. It is important to note that not all planned wind farms will eventually be built, thus we are assessing a worst-case-scenario. On the other hand, it also must be noted that there will be also be new territories that are not yet known, which potentially biases the estimate downwards. Here, we do not make projections about the future white-tailed eagle population but use as a reference population to assess its exposure towards wind turbines (reference population = territories that were active at least once between 2017 and 2022, $n = 712$).

2.6. Age structure and seasonality of wind turbine victims in Finland

White-tailed eagle wind turbine victims were found opportunistically between 2005 and 2022. We collected information on collisions from potential sources (i.e. Ringing center at Finnish Museum of Natural History, University of Helsinki), the most important being the ones from which the carcass and cause of death was examined by a veterinarian of the Finnish Food Authority (Isomursu et al., 2018). Age (five ages: 1st – 4th calendar year (cy) and adult) was determined based on plumage characteristics or was known if the bird was ringed; and season based on the month of discovery (winter: December, January, February; spring: March, April, May; summer: June, July, August; autumn: September, October, November).

3. Results

3.1. Apparent survival

We identified from the nest sites 742 adults, of which 438 were females and 304 were males. Of these, 360 were detected in multiple years (mean = 2.8, range = 2–9). When assessing their apparent annual survival, we found that wind turbine presence in a 5 km radius was the best model (Tables 1, S6) and the LRT indicated statistical significance ($P = 0.038$). The 6 km radius had similar parsimony and results (Table S7). For the top models, predicted survival estimates with and without a wind turbine in a 5 km radius around the nest site were 0.85 (SE = 0.04) and 0.92 (SE = 0.01), respectively (Fig. 2; Table S6). Resighting probability was higher for females than males, and higher if a nesting attempt was documented (Tables S6, S7). Resighting probability was independent of whether individuals were identified from DNA or photograph (Table S6). In a post-hoc analysis, we evaluated whether including wind turbine presence in a 5 km radius in resighting probability affected the top model on survival. Model results and parsimony were very similar to the original analysis. The top model included wind turbine presence in 5 km (AICc = 2872.27, ΔAICc = 2.48 from the best null model, Table S8). The reduction in survival persisted and estimates with and without a wind turbine in a 5 km radius around the nest site were 0.83 (SE = 0.04) and 0.92 (SE = 0.01), respectively. Resighting probability was 0.20 (SE = 0.05) and 0.21 (SE = 0.03) with and without a wind turbine in a 5 km radius, respectively. There was no violation of goodness-of-fit for the dataset (test2ct: $P = 0.353$; test2cl: $P = 0.871$; test3sm: $P = 0.736$, test3sr: $P = 0.408$) and median ĉ (0.91) did not indicate any overdispersion. Feather collection probability was independent of whether wind turbines are in the vicinity of a nest or not (see Supplementary material for details).

3.2. Breeding dispersal in relation to wind turbine presence

The data contained 481 reoccurring nesting events, including 96 dispersal events, in which white-tailed eagles moved to alternative nest sites (see detailed sample sizes Table S2). In case of breeding movement to an alternative nest, the median distance between nests was 1.00 km (mean = 6.01 km, SD = 12.39, max = 60.94 km). Eagles were not more likely to move to alternative nests if a wind turbine was present (Table 2a). The dispersal distance between nest sites was not farther but in contrast appeared shorter if there was a wind turbine within a 5 km radius than when there was no wind turbine (ΔAICc = 2.45; LRT: $\chi^2 = 4.70$, df = 1, $P = 0.030$; present, mean = 5.30 km, SE = 0.26; absent, mean = 7.34 km, SE = 0.93; estimate = −2.04, SE = 0.94; Table 2b).

3.3. Population-level mortality events and future predictions of wind turbine constructions

The first wind turbines within the 5 km radius of a territory were constructed in 1993. In 2014, 9 % of all known white-tailed eagle pairs have been affected by a wind turbine in this radius. The number of affected pairs remained constant at 8–9 % until 2018, when there was a drop to 5–6 % (Fig. 3). Active territories had 1–46 wind turbines within a 5 km radius in 2022 (median = 8). In a future worst-case-scenario there might be a rise from 28 (active in 2022, minimum) and 43 (active and inactive territories in 2022, maximum) to 92 territories being affected by 2030 (note that there is no minimum or maximum value as it is unclear what the ratio of active to inactive pairs will be).

Table 1
Top 10 Cormack-Jolly-Seber (CJS) models of annual survival probability of 742 white-tailed eagles recorded at Finnish nests between 2010 and 2022. Survival probability (Φ) was modelled with presence (absent = 0, present = 1) of wind turbines at varying radii (3–9 km) around nest locations and sex (factor, 'male' or 'female'). Resighting probability (ρ) was modelled as year (factor, 2010–2022), nesting outcome in a specific year (factor, 'inactive', 'active', 'successful', 'unknown'), sex, mean number of years the nest was visited by volunteers in the study period (numeric, 2–11), whether an individual was identified visually or by genotype (factor), and mean nest latitude and longitude (of all observations of an individual, numeric). A list of all models fit can be found in Table S3. AICc = second order Akaike Information criterion, ΔAICc = difference in AICc to top model, ω_i = AICc weight, K = number of parameters, −2lnl = likelihood.

Model		AICc	ΔAICc	ω_i	*K*	−2lnL
Apparent survival Φ	Resighting probability ρ					
Φ (5 km WT)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2869.88	0	0.14	22	2825.65
Φ (6 km WT)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2870.27	0.38	0.12	22	2826.04
Φ (5 km WT + sex)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2871.20	1.32	0.08	23	2824.95
Φ (5 km WT + latitude + longitude)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2871.23	1.34	0.08	24	2822.95
Φ (6 km WT + sex)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2871.45	1.57	0.07	23	2825.20
Φ (6 km WT + latitude + longitude)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2871.55	1.66	0.07	24	2823.27
Φ (1)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2872.16	2.28	0.05	21	2829.95
Φ (sex)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2872.97	3.09	0.03	22	2828.74
Φ (latitude + longitude)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2873.13	3.24	0.03	23	2826.87
Φ (5 km WT + sex + latitude + longitude)	ρ (sex + time + outcome + obs. type + nest visits + mean lat + mean lon)	2873.23	3.35	0.03	25	2822.93

When estimating an approximate number of wind-turbine-related mortality events for the breeder population, we found that wind turbine presence within 5 km of nest sites might have caused an additional total of 73.3 (active territories) – 97.5 (active and inactive territories, SD = 6.7) adult mortality events between 1991 and 2022 (Fig. S3). This represents an approximate 4.3–5.4 % mortality increase on the

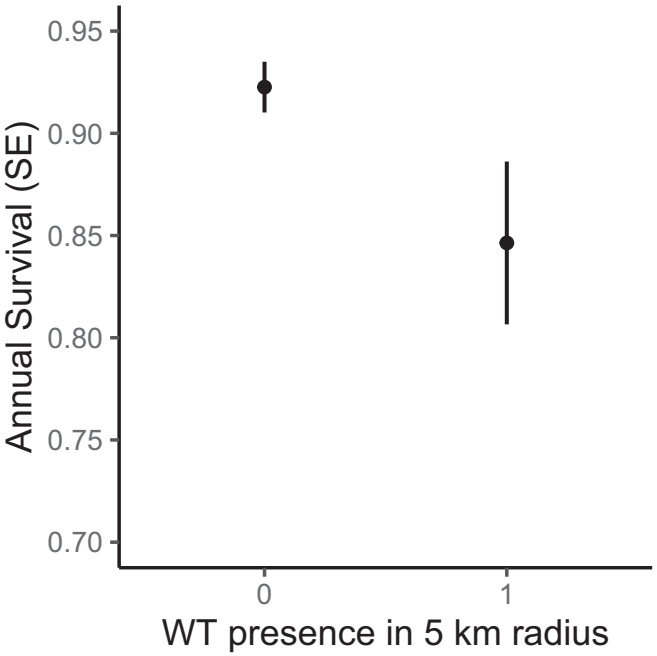


Fig. 2. Annual apparent survival rates (Φ) in relation to presence or absence of wind turbines (WT) in a 5 km radius. The top model for annual apparent survival included in addition to wind turbine proximity also resighting probability (ρ) with time (factor, 2010–2022), sex (factor, 'female' or 'male'), observation type (factor, 'visual' or 'genotype'), number of years a nest had been visited over the observation period (numeric, 2–11), mean latitude and mean longitude of all nests an individual had been seen (both numeric) and nest status in a year (factor, 'unknown', 'inactive', 'active' or 'successful nesting attempt'). Error bars depict standard error.

Table 2
Top 5 models of breeding dispersal, measured as movement to alternative nests in binomial GLMMs and top 5 models of breeding dispersal distance to an alternative nest of LMM. Data for the breeding dispersal probability model (a) contained 481 observations of individuals either remaining at the same nest in subsequent years (0) or moving to an alternative nest (1). Co-variables considered were sex (factor, 'female' or 'male), spatial position in the study area (nest latitude and longitude from the location of movement). Data for the breeding dispersal distance model contained 96 observations of breeding dispersal events of white-tailed eagles to alternative nest sites. Co-variates considered were sex, spatial position in the study area and years between sightings ('gap'). In all models random effects were year and individual ID. Variables of interest in both model comparisons were wind turbine presence in different radii (3–9 km) around the nest site in the year before moving. A list of all models fit can be found in Tables S4 and S5. AICc = second order Akaike Information criterion, ΔAICc = difference in AICc to top model, ω_i = AICc weight, K = number of parameters, −2lnl = likelihood.

Model	AICc	ΔAICc	ω_i	*K*	−2lnl
Breeding dispersal probability, $n = 481$					
1	477.87	0.00	0.09	3	−235.91
latitude + longitude	478.36	0.48	0.07	5	−234.11
4 km WT	478.70	0.83	0.06	4	−235.31
3 km WT	478.99	1.12	0.05	4	235.45
4 km WT + sex + latitude + longitude	479.04	1.17	0.05	6	233.43
Breeding dispersal distance, $n = 96$					
5 km WT + sex + gap	397.92	0.00	0.17	7	−191.32
5 km WT + gap	398.24	0.32	0.15	6	−192.65
4 km WT + gap	398.80	0.88	0.11	6	−192.93
4 km WT + sex + gap	399.15	1.24	0.09	7	−191.94
sex + gap	400.37	2.45	0.05	6	−193.72

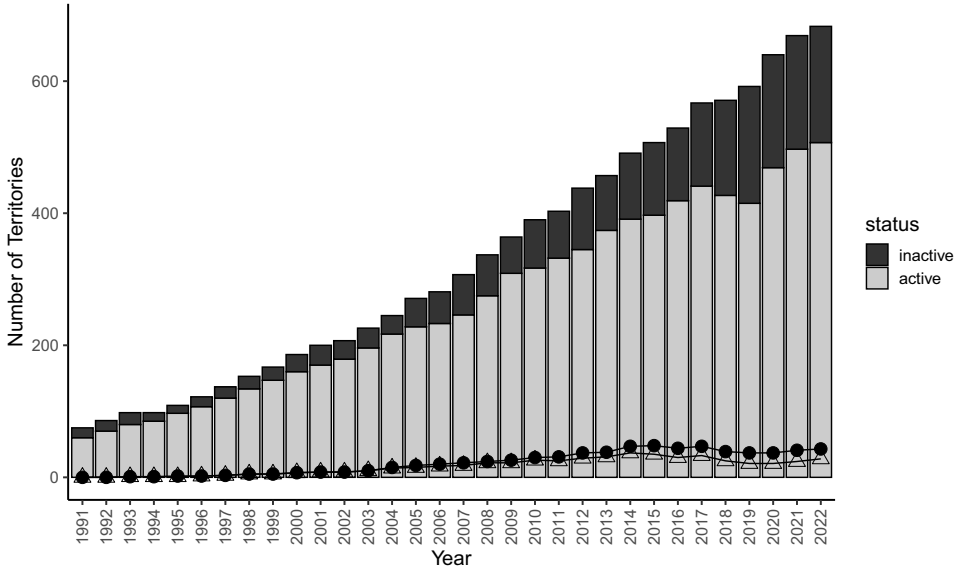


Fig. 3. Development of the white-tailed eagle population (number of territories) in the Finnish Baltic Sea coast, and number of territories containing a wind turbine within a 5 km radius. Grey barplots: active territories, dark grey barplots: inactive territories (i.e. no active nest found in the year in question but territory potentially still existing). The Finnish Baltic Sea population has grown since 1991 from 75 (including 15 inactive) to 683 (including 176 inactive) territories in 2022. Grey triangles indicate the number of active territories with wind turbines in a 5 km radius, solid circles the total number of territories (including inactive ones) with wind turbines in a 5 km radius. The proportion of territories with a wind turbine within a 5 km radius ranges from 0 % in 1991, to 9.5 % in 2014 and 6.3 % in 2022 (including inactive territories) and 5.5 % (active territories).

population-level. The estimate of total mortality events of adult breeders, including estimates of wind-turbine-related mortality, is 1506 (active territories, minimum value) – 2190 (active and inactive territories, maximum value) on the population-level. Current annual wind turbine-related mortality is estimated to be 4 (active) – 6 (total population) breeder mortality events (with high uncertainty: SD = 6.3;

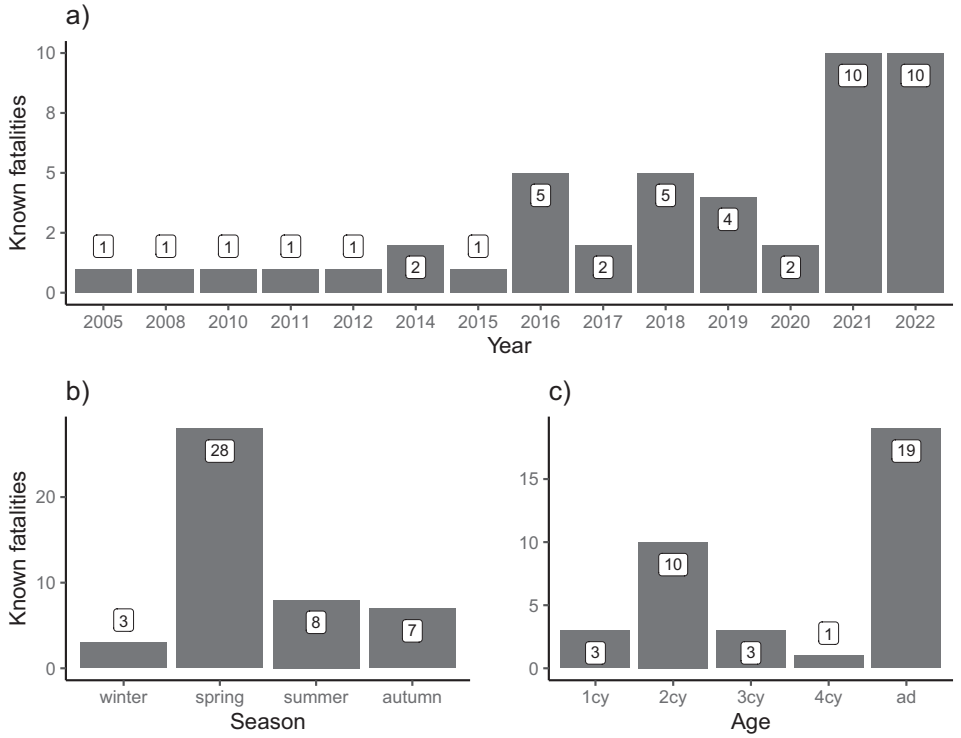


Fig. 4. Temporal trend (a), seasonality (b) and age-structure (c) of 46 white-tailed eagle fatalities found in Finland. Search effort was conducted unsystematically, thus interpretability is limited. (a) Temporal trend of 46 known white-tailed eagle fatalities due to wind turbines in Finland. There is a noticeable increase in mortality with the increase of wind turbines constructed mid-2010s. (b) Seasonality of carcass recoveries (winter: December, January, February; spring: March, April, May; summer: June, July, August; autumn: September, October, November) show a peak in spring. (c) Age structure (1st calendar year (1 cy) – 4th cy: first – fourth calendar year; ad: adult, 5th cy or older). Most eagles recovered are 1st cy birds and adults (older than 4 cy). Sample sizes are given in white boxes inside or above the barplot.

Fig. S4).

3.4. Age structure and seasonality of wind turbine fatalities in Finland

In total, 46 white-tailed eagle opportunistically found and reported carcasses were recorded from under wind turbines between 2005 and 2022. Information on their age was available for 36 and the recovery month from 46 individuals. Although the data was not systematically collected, there is indication that there have been more fatalities since mid-2010s (Fig. 4a), and more carcasses have been collected in spring than in other times of year (Fig. 4b). Adults and second calendar year birds appear to be most common victims (Fig. 4c).

4. Discussion

We utilized a large-scale dataset gathered from Baltic Sea coastal regions in Finland, to quantify the impact of wind turbines on a large soaring raptor, the white-tailed eagle. Our findings indicate a reduction in annual adult survival rates in the presence of wind turbines. The reduction in survival was best seen for wind turbine proximity within a 5 km radius. Current expected wind turbine-caused mortality events appear low on the population-level, but even slight decreases in vital demographic rates can have quite considerable impacts on long-lived and slowly reproducing species, such as the white-tailed eagle.

4.1. Lower survival in wind turbine proximity and potential population-level consequences

The presence of wind turbines within 5 km from the nest site was linked to a measurable decline in mean annual apparent survival from 92 % to 85 %, representing a 7.6 % decline. While survival estimates for unaffected eagles are associated with high certainty, there is more uncertainty associated with individuals affected by wind turbines. This is likely due to the smaller sample size of this group, but there can also be considerable variation between individuals and sites. The found decline is consistent with other research that has linked wind turbine proximity to a decrease in reproductive success in white-tailed eagles on Smøla (Dahl et al., 2012) and in Finland (Balotari-Chiebao et al., 2016b), but also in other raptor species (Fernández-Bellon et al., 2015; Kolar and Bechard, 2016) and even passerines (Mahoney and Chalfoun, 2016). Nonetheless, during the observed period, breeding white-tailed eagles exhibited high survival rates if unaffected by wind turbines. This value is higher than previous estimates for adults (combined for breeder and non-breeder) of this population (Nebel et al., 2023; Saurola et al., 2003), but lower than estimates from Scotland and Sweden ($\varphi = 0.94$–0.99, Evans et al., 2009; Green et al., 1996; Helander, 2003).

We had limited observations of repeated nesting attempts, particularly in the presence of wind turbines ($n = 18$ with wind turbine within a 5 km radius). However, there was no significant indication of a behavioural response to disturbance as we did not find a higher breeding dispersal probability to move to alternative nest sites if a wind turbine was present. If a breeding dispersal did occur near turbines, it happened over significantly shorter distances compared to areas without wind turbines, suggesting that the observed dispersal was more likely to be within the established territory than due to disturbance causing territory abandonments. Usage of alternative nest sites within the same territory can be a natural response to environmental cues, reduction of ectoparasite exposure (Ontiveros et al., 2008), or pairs might build nests in their territory as a response to nest failure (Postupalsky, 1974).

In recent years, both wind power and the white-tailed eagle population have grown considerably (Finnish Wind Power Association, 2023; Högmander et al., 2020), but the percentage of territories situated in the vicinity of wind turbines has not grown at the same rate. This can mainly be explained by new wind farms being built in inland regions instead of placing them directly along coastlines in recent years (Finnish Wind Power Association, 2023). However, several wind farms are currently in the planning stage close to the coast (Finnish Wind Power Association, 2023) that would be located within 5 km of a white-tailed territory. Especially when considering potential impact of wind turbines on other age classes that might be more affected due to ranging over larger distances (see i.e. 10 % decline in survival in white-tailed eagle juveniles born near the Smøla wind farm, Nygård et al., 2010), the wind power expansion as it is currently planned might be a threat for white-tailed eagle conservation with large-scale impacts on the Baltic Sea white-tailed eagle population in Finland.

The decrease in apparent survival observed for white-tailed eagles due to wind turbine proximity appears quite moderate in comparison to other large, soaring raptors. Previous research has shown that raptors face varying collision risks at wind farms due to differences in behaviour (Diffendorfer et al., 2021; Thaxter et al., 2017). Large vultures are especially vulnerable (decreases in survival by 15–30 %, Martínez-Abraín et al., 2012; Monti et al., 2022). Territorial Egyptian vultures *Neophron percnopterus* (Carrete et al., 2009) and golden eagles (Grainger Hunt et al., 2017) showed no measurable decrease in survival (despite them being collision victims, i.e. Ferrer et al., 2012; Katzner et al., 2017). Beside species-specific differences, age also plays a considerable role when assessing collision risks. Although survival of territorial golden eagles were found to be unaffected, non-breeders experienced a considerable decline in survival and local populations affected by wind turbines might only be maintained by immigration (Grainger Hunt et al., 2017). Golden eagles are also globally considered a species at risk due to wind power expansions (Balotari-Chiebao et al., 2021; Diffendorfer et al., 2021; Katzner et al., 2017). Sex-biased collision risk was found in several species (e.g. Heuck et al., 2020; Lane et al., 2020; Morinha et al., 2014), but this is not universal across species (Santos et al., 2021). Conserving vulnerable species requires understanding their behaviour and associated risks. While studying every species in detail is not feasible, the high mobility of soaring raptors, including territorial individuals, suggests that similar conservation strategies used for protecting the white-tailed eagle may be applicable. Lessons learned from studying the white-tailed eagle can help guide the conservation of other species facing similar challenges.

In our combined modelling approach, we tried to account for alternative mechanistic explanations for the found lower apparent survival rates, including replacement in the form of breeding dispersal. This required the eagle to settle at a new nest and the discovery of the nest by our volunteers. Finding new nests requires high search effort, and new nests might go undetected for multiple years, therefore providing some level of bias. Only more intense long-term monitoring would help to increase the certainty of our estimates.

4.2. Age-structure and seasonality of wind collision fatalities

Although our opportunistic data on collision fatalities only allows limited insight into collision risk (supplementary material), our results partly match with findings from other, systematically collected studies. Spring might be associated with times of high collision risk for white-tailed eagles due to increased amount of wind and updrafts, but also territorial disputes (c.f. May et al., 2010). The predominance of adult individuals among the collected carcasses might indicate high number of floaters in adult-plumage the Finnish population or increased collision risk.

4.3. Management and conservation recommendation

We recommend four main conservation actions: (1) carrying out continuous environmental assessments and monitoring, (2) establishing a larger, >5 km safety buffer, which would encompass protection of breeding location and higher proportion of foraging areas of breeders than the current 2 km buffer guideline used in Finland, (3) implementing mitigation strategies, and (4) develop environmental impact assessments and planning tools (e.g. habitat models and cumulative impact

assessments) so that both territorial and population impacts can be better assessed in the future. While larger safety buffers would primarily protect breeding white-tailed eagles, mitigation by different rotor blade designs (e.g. May et al., 2020, 2012) or automatic detection of eagles and stopping of turbines in their presence might also protect non-breeding white-tailed eagles and other species (Balotari-Chiebao et al., 2021). Continuous monitoring of collision events would be necessary to monitor collision mortality and adapt conservation actions for various species in the Baltic area and Finland, as they are currently lacking. Moving new construction sites to more inland habitats or offshore appears to be the most effective conservation strategy regarding white-tailed eagles. However, this approach may impact other vulnerable species (e.g. golden eagles, various grouse species). Therefore, finding an optimal strategy requires carefully weighing the costs and benefits to ensure the preservation of sensitive species alongside renewable energy development.

5. Conclusions

Quantifying demographic parameters of elusive and hard-to-detect raptor species is a challenge in their conservation (Watson et al., 2018). By using large-scale data and non-invasive identification method, we found indication that white-tailed eagles in the Baltic Sea coastal population might be negatively impacted by wind turbine proximity, although small sample sizes do not allow confident conclusions. Impactful distances between breeding raptors and wind turbines can be much larger than is often considered (Husby and Pearson, 2022; Krone and Treu, 2018). Impacts on other white-tailed eagle age classes and the population-level consequences on a nationwide scale, and how the future expansion of wind power in the Baltic area will affect the population, is unclear at this point and should be focus of future studies. Insights gained from studying the white-tailed eagle can inform the conservation of other species.

CRediT authorship contribution statement

Carina Nebel: Writing – original draft, Visualization, Validation, Methodology, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Torsten Stjernberg:** Writing – review & editing, Validation, Resources, Data curation. **Hannu Tikkanen:** Writing – review & editing, Resources, Formal analysis, Data curation. **Toni Laaksonen:** Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Methodology, Funding acquisition, Data curation, Conceptualization.

Declaration of competing interest

We declare that we do not have any competing interests.

Data availability

The data that has been used is confidential.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2024.110604.

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