Biological Conservation 292 (2024) 110509

Contents lists available at ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Monitoring carcass persistence in windfarms: Recommendations for estimating mortality

Andreas Ravache[a,b,*], Kévin Barré[a,b], Brice Normand[c], Corentin Goislot[c], Aurélien Besnard[d], Christian Kerbiriou[a,b]

[a] Centre d'Ecologie et des Sciences de la Conservation (CESCO), Muséum national d'Histoire naturelle, Centre National de la Recherche Scientifique, Sorbonne Université, Paris, France
[b] Centre d'Ecologie et des Sciences de la Conservation (CESCO), Muséum national d'Histoire naturelle, Centre National de la Recherche Scientifique, Sorbonne Université, Station Marine, Concarneau, France
[c] Bureau d'études Ouest Am' SCOPE-SA, 1 rue des Cormiers, 35651 Le Rheu, France
[d] CEFE, Univ Montpellier, CNRS, EPHE-PSL University, IRD, Montpellier, France

ARTICLE INFO

Keywords:
Environmental impact assessment
Carcass removal trials
Mortality estimate
Wind turbine
Carcass persistence
Survey effort

ABSTRACT

Environmental impact assessments (EIAs) of windfarms require counting bat and bird carcasses around turbines to assess the number of individuals killed by collision or barotrauma. In order to accurately estimate mortality from carcass counts, most studies correct mortality by the probability of carcass removal between fatality searches, data that is obtained through persistence trials. However, there is currently no consensus on persistence trial design and estimation methodology. This study aimed to assess the sensitivity of several estimators employed during EIAs to collect persistence data. To this end, the persistence time of 266 carcasses was surveyed continuously using camera traps. This continuous observation data was resampled to generate virtual datasets collected using different survey intervals. Several estimators were then used to estimate mortality from a sub-sample of carcasses for each carcass survey interval. Mortality estimates obtained by the different estimators were very similar when persistence was observed continuously. However, differences between estimators emerged when persistence survey intervals widened, and even more so when fatality searches were conducted less frequently. The GenEst estimator seemed to produce less biased and more accurate mortality estimates compared to other estimators frequently used for EIAs. Based on these findings, we recommend limiting search intervals during persistence surveys as much as possible. If persistence time is low, we also recommend increasing fatality search frequency. For estimators using mean persistence time, the latter should be calculated using the median time between the last observation of the carcass and its first absence.

1. Introduction

In the context of the transition to a larger share of decarbonized energy, windfarms represent a primary contributor to clean energy production. Wind energy is predicted to supply more than one-third of global needs by 2050 (IRENA, 2019). Over the past few decades, wind energy installations have grown rapidly, with production increasing by 10–20 % per year (IRENA, 2019; Kumar et al., 2016). Although wind energy has a small carbon footprint, it is not exempt from environmental impacts. Studies have revealed direct effects on biodiversity, such as bat and bird mortality related to collisions or barotrauma (Carrete et al., 2009; Erickson et al., 2014; Frick et al., 2017; Schippers et al., 2020), as well as indirect effects such as behavioral disturbance, loss of habitat use or trophic cascade effects on a wide variety of taxa (Barré et al., 2018; Horn et al., 2008; Leroux et al., 2022; Millon et al., 2018; Scholz and Voigt, 2022). Direct mortality of bats and birds can have a significant, though little studied, impact on populations (Carrete et al., 2009; Duriez et al., 2023; Frick et al., 2017). As the adverse impacts of windfarms on biodiversity are likely to increase with the new installations planned to meet clean energy production goals, a comprehensive and accurate assessment of these impacts is increasingly necessary. This will provide useful information for the spatial planning of future wind parks and for impact reduction measures.

Following the installation of windfarms, some countries require bird

* Corresponding author at: 7 rue Theophile Louarn, 29900 Concarneau, France.
E-mail address: andreas.ravache@mnhn.fr (A. Ravache).

https://doi.org/10.1016/j.biocon.2024.110509
Received 5 September 2023; Received in revised form 5 February 2024; Accepted 15 February 2024
Available online 21 March 2024

and bat carcasses resulting from collisions to be counted ("fatality searches") in a defined area around turbines in order to estimate windfarm-related mortality, as part of the environmental impact assessment (EIA) procedure (e.g. Scotland: SNH, 2009; Portugal: APA, 2010; France: MTES, 2018; Spain: MTERD, 2020). However, there is a risk that the number of carcasses detected during fatality searches may underestimate the actual mortality due to two main sources of bias: searcher efficiency and carcass persistence (Barrientos et al., 2018; Smallwood, 2007). Searcher efficiency is the probability of a searcher to find carcasses that are present in the field, while carcass persistence is the probability that a carcass has not been removed by a scavenger or other source before the fatality search was conducted. Adjusting for these two factors alone can increase the mortality estimate by a factor of up to 40 compared to the raw number of carcasses found (Smallwood, 2007). In order to account for these biases, searcher efficiency and carcass persistence are often estimated through specific experiments conducted at the monitoring site as part of the EIA (known as "carcass detection trials" and "carcass removal trials" or "carcass persistence trials", respectively). Searcher efficiency is estimated by randomly placing carcasses under turbines and then assessing the searcher's ability to find these. The average persistence time (i.e. the time between carcass deposit and its disappearance) is estimated by placing a set of carcasses under turbines and monitoring their presence over a specific period. It is then used to calculate the average persistence probability between fatality searches (e.g. Bispo et al., 2013a; Costantini et al., 2017). The estimates of persistence and searcher efficiency are subsequently employed in estimators to yield corrected mortality estimates (Dalthorp et al., 2018; Erickson et al., 2004; Huso, 2011; Korner-Nievergelt et al., 2011; Péron, 2018; Rabie et al., 2021). The accuracy and bias of these estimates can significantly impact the final conclusions of the EIA and subsequent decision-making, and are therefore of critical importance (Huso, 2011; Korner-Nievergelt et al., 2011).

The mean persistence time is often calculated at the time Mean carcass persistence time can vary greatly depending on the site (e.g. ranging from 1.2 days to 25.8 days between studied sites in Italy: Costantini et al., 2017), and can be influenced by season, location or surrounding habitat (AWWI, 2020; Barrientos et al., 2018). Additionally, the frequency of carcass surveys and the way mortality estimates are derived are two factors that can affect the accuracy and bias of mortality estimates. More frequent surveys of carcass presence should lead to more accurate and less biased estimates of persistence time, but have a higher financial cost. Understanding how survey effort may affect the accuracy of persistence estimates is thus crucial to provide recommendations for the implementation of standardized protocols. The required interval time between searches in order to accurately estimate carcass persistence is not well established, and varies between studies (Smallwood, 2007). Despite the existence of national guidelines, these recommendations are usually not compulsory, and most protocols leave considerable flexibility in the frequency of surveys (e.g. in France: MTES, 2018). This can lead to variation in the bias and accuracy of persistence estimates, and can therefore have an impact on mortality estimates and the resulting decisions.

Several methods for estimating persistence exist (Arnett et al., 2008). Persistence can be estimated by methods such as known-fate analysis (e.g. Villegas-Patraca et al., 2012), nonparametric maximum likelihood estimation of the survival curve (e.g. Kitano et al., 2020) or parametrical analysis based on several distribution families (e.g. Bispo et al., 2013a). Some national protocols recommend the use of simple correction formulas to estimate mortality from field observations (e.g. MTES, 2018). However, due to the right-censored (i.e. the removal event did not occur during the trial) and interval-censored (i.e. the removal time is only known to lie within a certain interval) nature of persistence trials, these formulas may not fully capture the complexity of carcass persistence over time (Leung et al., 1997). Other guidelines recommend the use of recently developed estimators (e.g. GenEst estimator, Dalthorp et al., 2018; MTERD, 2020), allowing the use of more complex survival analyses, survey schedules, and other carcass-related parameters, in an attempt to improve the accuracy of estimates. However, little is known about the ability of these different estimators to handle variable survey intervals of carcass persistence.

In this context, this study assessed the sensitivity of several estimators employed during EIAs of windfarms in order to better design persistence data collection. The objectives were (1) to study the impact of carcass persistence survey intervals on mortality estimates according to different estimators in different scenarios of fatality search schedule and observed mortality, (2) to compare the reliability of mortality estimates produced from a reduced sample of carcass observations obtained locally with estimates obtained from a larger dataset, and (3) to use the results to provide recommendations for defining more efficient EIAs. Prior to the study, we hypothesized that (i) wider survey intervals would lead to less precise estimations and (ii) more complex estimators incorporating the interval- and right-censored nature of field data would be less sensitive to variability in persistence survey intervals.

To assess this, we continuously monitored the removal of carcasses in the vicinity of wind turbines using camera traps. Continuous observation data was then resampled to generate virtual datasets collected using different survey intervals. After subsampling a realistic number of carcasses monitored during persistence trials from resampled datasets, we estimated mortality rates with several estimators for each carcass survey interval, and for different scenarios of number of carcasses found and fatality search intervals.

2. Materials and methods

2.1. General workflow

The workflow we designed (Fig. 1) aimed to evaluate the accuracy and bias of different methods used to estimate carcass persistence and mortality rates during EIAs of windfarms, depending on the frequency of surveys, under various scenarios of the number of carcasses and fatality search intervals. In the first step (1), we continuously observed the persistence of carcasses in the vicinity of wind turbines using camera traps. (2) The second step aimed to visually identify removal events from camera trap images. (3) In the third step, the removal event data was resampled to mimic a lower frequency of carcass persistence surveys, ranging from one survey per day to one survey every five days. To mimic realistic and mandatory monitoring, which depends on working days and guidelines, a semi-random survey was also simulated. Continuous observations were kept in the analyses. (4) In the fourth step, random samples of 15 out of the 218 monitored carcasses were drawn for each survey frequency from the continuous and the resampled datasets. (5) Carcass persistence was then estimated in the fifth step, using three different estimators for each of the resampled datasets. (6) The sixth step aimed at estimating mortality rates for each estimator and carcass survey frequency using simulated parameters for fatality search intervals and number of carcasses found. (7) Finally, in the seventh step, the mortality estimates obtained were compared to the reference value of each estimator, obtained by estimating carcass persistence from the raw continuous carcass persistence survey.

Steps (4) to (7) were repeated 1000 times, from random sampling to comparison to the reference value. Different scenarios were tested with this workflow using different combinations of fatality search intervals (3, 7 or 14 days) and of number of carcasses found (5 or 10 carcasses), resulting in a total of six scenarios.

2.2. Camera trap persistence trials

Carcass persistence trials were conducted between 2021 and 2022 at 16 windfarms located in western France. A total of 266 carcasses were placed in the vicinity of wind turbines. The number of carcasses per windfarm varied from 1 to 15 (10.2 ± 4.8). A maximum of four carcasses were placed simultaneously under a wind turbine to avoid scavenger

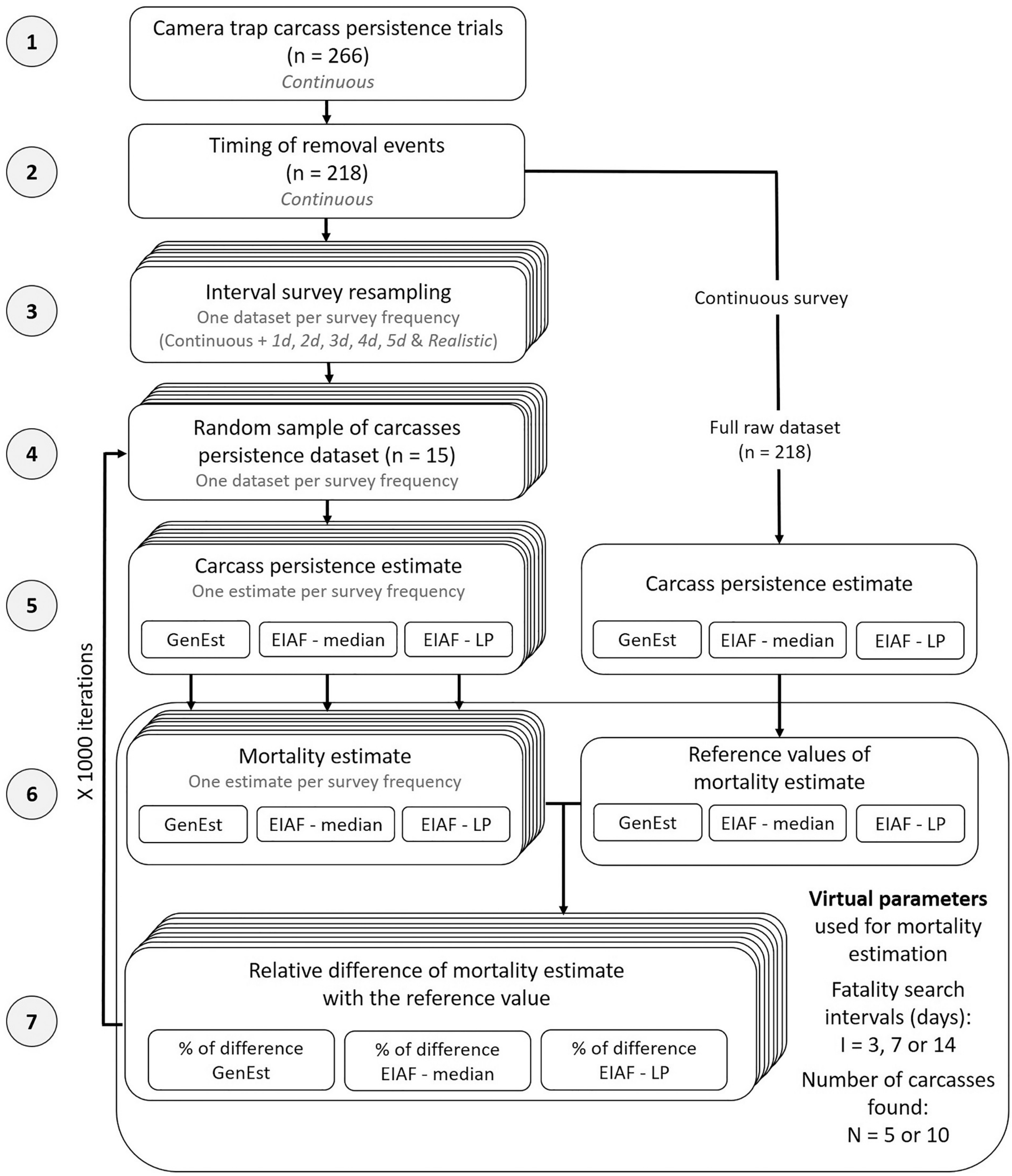

Fig. 1. General workflow to evaluate the accuracy and bias of different estimators used to estimate carcass persistence and mortality rates in relation to the carcass persistence survey frequency.

swamping (Barrientos et al., 2018; Smallwood, 2007), within a 50-m radius corresponding to the area surveyed during a fatality search (MTERD, 2020; MTES, 2018). Carcasses used in trials consisted of common mice (*Mus musculus*, 10–20 g), chickens (*Gallus gallus*, 30–40 g) or common wood pigeons (*Columba palumbus*, 380–420 g). We chose these species because they are the most commonly used during persistence trials, as observed from a collection of >200 reports gathered in France (author personal observation). However, it should be noted that the use of such carcasses may not always reflect the persistence of wild animals (Smallwood, 2007; Urquhart et al., 2015). All carcasses were purchased in animal stores.

Digital infrared camera traps (Boly® SG520 HD SCout Guard) equipped with 32GB SD cards were used to monitor the persistence time of each carcass. Camera traps were placed at a distance of 1 to 3 m from the carcasses, fixed on stakes at a height of about 80 cm, and oriented toward the carcasses to be monitored. Cameras were programmed to take a burst of two pictures when motion was detected, with a minimum interval of 5 min between two consecutive bursts to avoid memory saturation. Camera sensitivity was set to “low” to avoid the detection of movements such as wind-blown grass. Timelapse photos were also taken every 30 min to estimate the removal of the carcass even if motion detection did not work. Cameras were checked 7 days after their placement to ensure that the battery was not depleted and that the SD card was not full. Cameras were removed after 14 days even if carcasses were still present.

2.3. Timing of removal events in the continuous survey

Camera trap pictures were meticulously examined to determine removal events as precisely as possible. The images treatment resulted in a data frame consisting of a row per carcass. For each carcass, we calculated the time elapsed between the deposit of the carcass and its last observation on the camera (lower limit). Additionally, we computed the time between deposit and the first observation of its absence (upper limit) if the removal event occurred during the survey period (i.e., 14 days). These limits will later serve as lower and upper bounds for interval-censored survival analyses. If a carcass persisted beyond the survey period, the removal event was considered right-censored.

Most of the time, the scavenger was visible in the camera trap images, and removal events were identified with a precision of a few seconds. Given that the study aimed to investigate the methodology and intervals used to estimate persistence duration, only removal events identified within an interval of less than one hour were included in the analyses. Cameras that ceased functioning before the end of the 14-day survey were also excluded from the analyses. Different images from camera traps, representing various cases of removal event identification or misidentification, are presented in Fig. S1 (see Supplementary material).

2.4. Carcass persistence survey interval resampling

The precise timing of removal events (i.e. a disappearance event for which the time is known with an accuracy of less than one hour) captured by the camera traps during the continuous survey was resampled to mimic lower carcass persistence survey frequency (Fig. 2). To this end, lower and upper limits of the removal event intervals were rounded to mimic carcass persistence survey frequency ranging from daily surveys to surveys every five days, as well as intervals based on a mandatory survey schedule, taking into account EIA recommendations on monitoring frequency and the absence of monitoring on weekends (e.g. MTES, 2018). The mandatory persistence survey intervals were set as follows, with the carcass placement day being day 0: one check at day 1, one randomly selected check between day 2 and day 4, one randomly selected check between day 5 and day 8, one randomly selected check between day 9 and day 12, and one check on day 14 (Fig. 2). This resampling resulted in seven carcass persistence datasets (continuous observation by camera trap, 1 day, 2 days, 3 days, 4 days, 5 days and mandatory survey), each representing the 218 interval-censored carcass removal events, but observed within intervals of different lengths. A visual representation of the lower and upper bounds of carcass persistence for each survey interval is presented in Fig. S2 (see "Supplementary material").

2.5. Random sampling of carcass observations

A random selection without replacement of 15 carcasses, representing a conventional number of carcasses monitored during persistence trials (APA, 2010; MTERD, 2020; MTES, 2018), was drawn from each resampled carcass removal dataset corresponding to each survey interval.

2.6. Carcass persistence probability

Persistence estimates were then estimated from these subsamples using three different methods, as described below. These three methods of persistence estimation (GenEst, EIAF-median and EIAF-LP) were used to estimate carcass persistence, resulting in three carcass persistence estimates for each of the seven survey intervals considered.

2.6.1. GenEst persistence estimate

Carcass persistence was first estimated using the "cpm" function implemented in the GenEst package (Dalthorp et al., 2018), using the lower and upper bounds of removal events. The "cpm" function fits a survival model to estimate the probability of persistence over time from interval-censored and right-censored observation data. Carcass persistence models were fitted with a log-logistic distribution family, which had the lowest AICc compared to exponential or Weibull families in our dataset (exponential = 2650.75; Weibull = 2628.93; log-logistic = 2603.52). The size of carcasses, season or other parameters potentially influencing carcass persistence were not considered in the persistence model to maintain consistency across estimators. Despite the existence of a user interface, the entire procedure was directly coded in R to provide more flexibility.

2.6.2. EIA formula persistence estimate

One of the most widespread estimates of the probability of carcass persistence used in EIAs is based on a formula derived and simplified from Huso (2011). This study provides an example, assuming that animals enter the morbid population at a constant rate during the interval between searches, and that persistence time follows an exponential distribution, which is the simplest persistence time model. The probability of carcass persistence is then described as:

$$P = t \times \left(\frac{1 - \frac{exp^{-I}}{t}}{I} \right)$$

Interval	Day 0	1	2	3	4	5	6	7	8	9	10	11	12	13	14
Continuous	P														
1 day	P	I	I	I	I	I	I	I	I	I	I	I	I	I	I
2 days	P		I		I		I		I		I		I		I
3 days	P			I			I			I			I		I
4 days	P				I				I				I		I
5 days	P					I					I				I
Mandatory	P	I	1 random			1 random				1 random					I

Fig. 2. Visual representation of carcass persistence survey interval resampling. P represents the placement of the carcass during the persistence trial at day 0. The first row ("continuous") represents the continuous survey of carcasses using the camera traps. Subsequent rows represent the different resampled survey intervals. Each bar represents a carcass check. The last row represents the mandatory survey, characterized by a check at day 1, 3 checks at random times, and a check at day 14.

with

P: average probability of persistence of a carcass
t: estimated mean persistence time in days
I: mean interval between fatality searches in days.

The Huso estimator was made publicly available through an R-based software in 2012, accompanied by a user guide that was revised in 2018 (Huso et al., 2018). The latter requires the user to indicate whether the observed data represent known persistence time, interval censored, or right censored data. The software then uses a maximum likelihood-based method to calculate estimated persistence time (t) for an exponential model as well as for other distribution families. The likelihood-based approach represents the most effective method for handling censoring problems, as it employs estimation methods that adjust for whether an individual observation is censored or not (Turkson et al., 2021).

Some national protocols recommend and reference the Huso formula for estimating the probability of persistence (e.g. APA, 2010; MTES, 2018). These protocols incorrectly adopt the formula presented as an example in Huso (2011) and described above as they do not provide explicit guidance on estimating the mean persistence time (t), which requires appropriate calculations in the presence of right-censored or interval-censored data. Discussions with consulting firms implementing this method indicated that two approaches are used to estimate this parameter. The first method defines the mean persistence time as the average of time elapsed between carcass placement and the median time between last observed presence and first absence of the carcass. When data is right-censored, the last presence is used in the average calculation instead of the median. This method is referred to hereafter as the **EIAF-median** (i.e. EIA formula - median). The second method defines mean persistence time as the average of time elapsed between carcass placement and last observed presence of the carcass. This method is referred to hereafter as the **EIAF-LP** (i.e. EIA formula - last presence").

2.7. Simulation of mortality rates from persistence estimates

The carcass persistence estimates were then used to estimate mortality rates. The persistence probability model obtained with the GenEst estimator was used to estimate fatality rates through a mortality estimation model (Dalthorp et al., 2018; Simonis et al., 2018). The EIAF-LP and EIAF-median persistence estimates were used in the formula derived from (Huso, 2011) and used in EIAs (APA, 2010; MTES, 2018). Since our study focused on the impact of persistence monitoring frequency, the other information required to estimate mortality was held constant, as described in each estimator's specific section. Several simulated parameters of fatality search intervals (i.e. the time elapsed between consecutive fatality searches under a turbine) and number of carcasses found under the turbines were tested to assess the variation in bias and accuracy of the estimators in different scenarios.

2.7.1. Simulation of mortality estimates using GenEst

Within the GenEst estimator, a mortality estimation model was built using the "estM" function. This uses the persistence probability model defined above to estimate mortality rate, along with a searcher efficiency model, a fatality search schedule, carcass observation data, density-weighted proportion and proportion of the facility surveyed (Dalthorp et al., 2018; Rabie et al., 2021; Simonis et al., 2018). The searcher efficiency model was defined by simulating the placement of 30 carcasses, all subsequently found by the observer (i.e. searcher efficiency = 1). The fatality search schedule was simulated by generating a regular sequence of dates between January 1 and December 30, every 3 days, 7 days or 14 days, according to the search interval scenario simulated. Carcass observations were simulated by randomly selecting, with replacement, 5 or 10 dates from the search schedule, representing the scenario of 5 or 10 carcasses being found. The density-weighted proportion model was set as 1, thus assuming that an animal killed by wind turbines had a 100 % chance of falling in the area prospected during fatality searches. The proportion of the facility surveyed was set as 1, indicating a prospection of 100 %. Then 1000 simulation draws were used to estimate mortality using the "estM" function, and the median estimate obtained was kept as the mortality estimate.

2.7.2. Simulation of mortality estimates using the EIA formula

In this simulation, mortality estimates were calculated using the following formula employed during EIAs of windfarms to estimate mortality from the number of carcasses found and correcting it using the probability of persistence estimated earlier, along with a surface correction coefficient, searcher efficiency estimate and the correction coefficient of the search interval (APA, 2010; Huso, 2011; MTES, 2018):

$$N = \frac{(Na - Nb)}{\left(d \times \frac{Min(I, -log(0.01) \times t)}{I} \times P\right)} \times A$$

with

Na: total number of carcasses found in the searched area
Nb: number of carcasses killed by something other than wind turbines
d: searcher efficiency
A: surface correction coefficient
t: mean persistence time in days
I: mean interval between searches in days
P: carcass persistence coefficient (defined above).

In line with the virtual parameters defined in the GenEst estimator, searcher efficiency and the surface correction coefficient were set to 1, meaning that all carcasses were detected during the persistence test and that the entire surface below the wind turbines were surveyed). The number of carcasses killed by something other than wind turbines to 0. The number of carcass observations was set as 5 or 10, and the interval between searches as 3 days, 7 days or 14 days, depending on the scenario simulated.

2.8. Number of iterations

The workflow—from random sampling of persistence events to the calculation of mortality estimates by the three methods—was repeated 1000 times. This resulted in 1000 mortality estimates for each carcass persistence survey interval and each estimator tested.

2.9. Reference value of mortality estimate

A reference value of the mortality estimate was estimated for each estimator using the full raw dataset of carcass persistence ($n = 218$) continuously observed with the camera traps. Other parameters required by the estimators were kept consistent with the other simulations to enable meaningful comparisons.

2.10. Bias and accuracy estimation of each method for each carcass survey interval

Finally, the bias and accuracy of each method and for each survey interval was obtained by calculating the relative difference of each estimate obtained through each iteration with the reference value of the estimators. The mean percentage of difference thus represented the bias of the estimator from the reference value, while the standard deviation of this difference represented its accuracy.

3. Results

3.1. Camera trap persistence trials and timing of removal events

Out of the 266 carcasses monitored using camera-traps, we did not encounter any instances where image inspection suggested a carcass had been removed, but an observer found the carcass still present. Thirty removal events (11.2 %) could not be dated due to cameras ceasing to function before the end of the 14 days survey, and 18 removal events (6.8 %) could not be timed with an accuracy of <1 h. Consequently, these data points were excluded from analyses. Among the 218 carcasses included in subsequent analyses, 168 were removed before the study's completion, indicating a removal rate of 77.1 % within 14 days. The scavenger responsible for the disappearance of the carcass was identified in 138 cases (82 %).

3.2. Mortality estimates

Overall, mortality estimates obtained by the different estimators were very close to each other when carcass persistence was monitored using camera traps (i.e. continuous survey, Fig. S3). As an example, mortality estimates obtained for the continuous persistence survey, under a scenario of 10 carcasses found and a fatality search interval of 7 days, were 19.1 ± 3.4 when using GenEst, and 18.4 ± 2.4 using EIAF-LP or EIAF-median. Mortality estimates produced by GenEst during the continuous survey were slightly higher for fatality search intervals of 3 or 7 days, and slightly lower for a fatality search interval of 14 days, compared to the results obtained using EIAF-LP or EIAF-median (Fig.

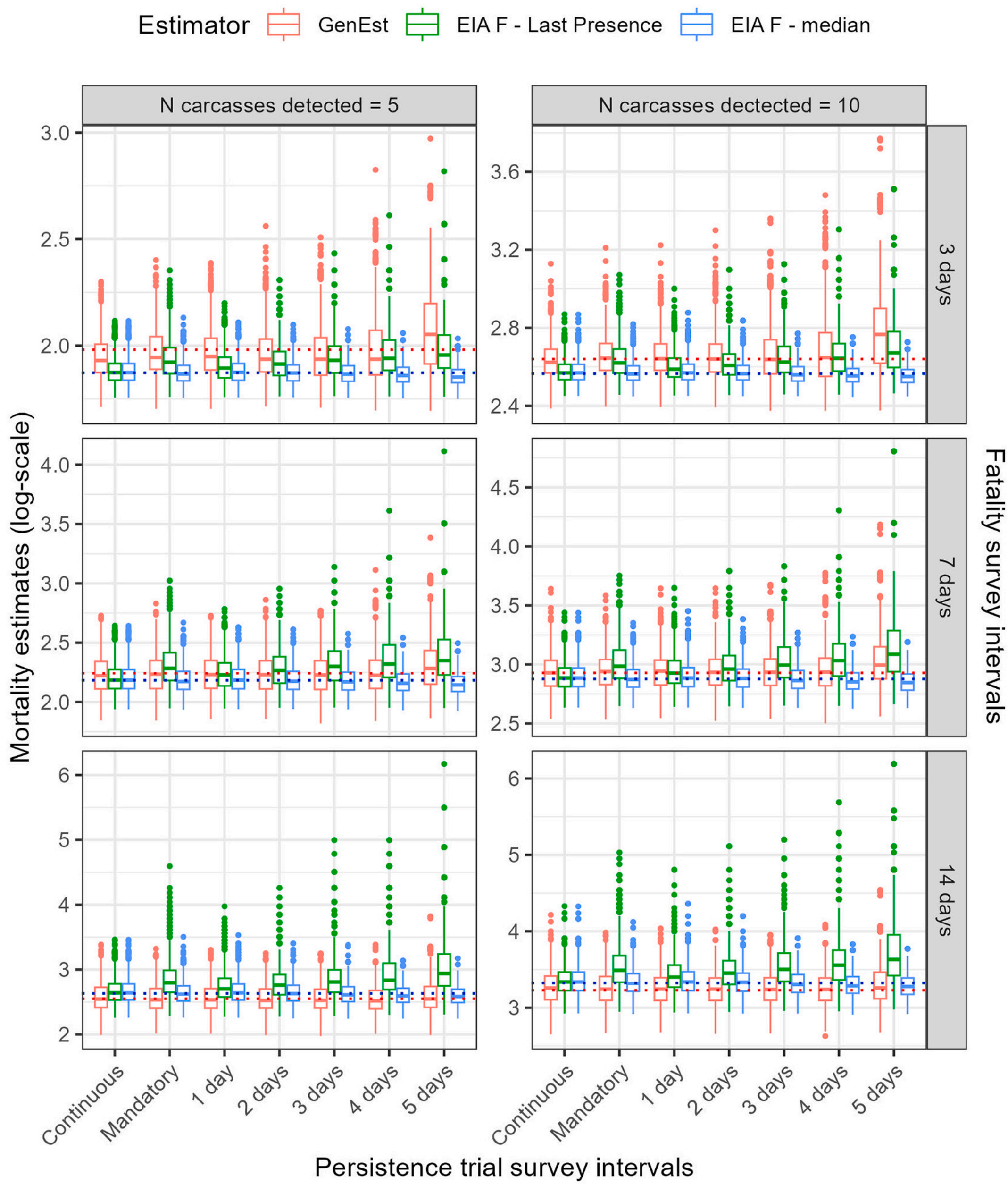

Fig. 3. Boxplots representing the mortality estimates on a log scale, obtained through each iteration using a subset of 15 random carcass observations for different persistence trial survey intervals (represented on the x axis of each panel), different scenarios of carcass observations (plot columns) and frequency of fatality searches (plot rows). The red dotted line represents the reference value calculated with GenEst with the entire persistence dataset of continuous carcass observations, and the blue dotted line represents the reference value similarly calculated using EIAF-median and EIAF-LP, both giving the same result. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

S3). However, differences between estimators increased for wider survey intervals. A same number of carcasses found during narrower fatality search intervals resulted in higher mortality estimates. Doubling the number of carcasses found during searches doubled the mortality estimate, but also impacted the standard deviation of the estimates. The GenEst and EIAF-median estimators produced similar estimates and were less sensitive to changes in persistence survey intervals compared to EIAF-LP. For a given estimator and persistence interval survey, the mean bias did not increase with an increase in the number of carcasses found, despite higher mortality estimates (Fig. 3 and Table 1). However, the bias of each estimator tended to increase when the intervals between searches were longer, more markedly for the EIAF-LP estimator.

3.2.1. *Mortality estimates obtained with GenEst*

The mean mortality estimates produced by GenEst using the subsamples of carcass persistence data were close to the reference value obtained from the full dataset of carcass persistence observed continuously. The mean bias was <5.9 % of the reference value in all scenarios using the mandatory carcass persistence survey intervals recommended for EIA (Table 1). Increasing persistence survey intervals up to 4 days did not strongly increase bias in mortality (bias range: [0.4; 7.3 %]), but significantly increased for a 5-day interval (bias range: [7.2; 18.4 %]). In addition, the accuracy decreased as the persistence survey interval lengthened, but this pattern was more pronounced for narrower fatality search intervals. Wider search intervals resulted in an increase in bias and a decrease in accuracy for most scenarios. Doubling the number of carcasses found resulted in slightly higher bias and lower accuracy (mean bias difference: 2.88 % and mean accuracy difference: 0.98 % across all fatality search and persistence survey intervals).

3.2.2. *Mortality estimates obtained with EIAF-LP*

The EIAF-LP method resulted in much higher bias from the reference value and lower accuracy of mortality estimates compared to the other two estimators. This estimator produced numerous outliers, estimating unrealistic values for mortality. Bias increased and accuracy decreased with the length of persistence survey intervals. These two parameters fluctuated even more as the interval between mortality monitoring became wider. For example, when fatality searches were conducted every 7 days and the persistence survey every 5 days, the estimated mean mortality differed by 35.3 % from the reference value, while the mean estimates differed by 92.1 % when fatality searches were conducted every 14 days. The number of carcasses found had a comparatively small influence on both the accuracy and bias of this estimator.

3.2.3. *Mortality estimates obtained with EIAF-median*

Calculating the mean persistence time using the median between the last presence and first absence of a carcass produced less biased and more accurate mortality estimates compared to the reference value when using the EIAF mortality estimation formula. Mortality estimates obtained with this estimator were also much closer to the results obtained with the GenEst estimator than those obtained with EIAF-LP. However, in contrast with GenEst, widening the persistence survey intervals led to a reduction in the mean mortality estimates, and a slight increase in accuracy. On the other hand, wider fatality search intervals resulted in a decrease in precision. The results were very similar for the two tested scenarios of number of carcasses found (5 or 10).

4. Discussion

4.1. *Interpretation of the results*

Overall, the bias and accuracy of the mortality estimates obtained from a subsample of carcass observation data differed from the reference value depending on the estimator. For a given scenario of number of carcasses found and fatality search interval, the mortality estimates were less biased and more accurate when persistence survey intervals were narrower, particularly for continuous monitoring. When carcass persistence was monitored by camera traps (i.e. continuous), all the

Table 1

Mean and standard deviation of the percentage of difference from the reference mortality estimates obtained by applying each estimator to the full persistence observation dataset. Red (almost invisible given the scale difference) indicates an underestimation of mortality estimates compared to the reference value, while green represents an overestimation. Blue bars represent the standard deviation of the persistence estimates. The length of the colored bars was scaled so that the bar size is a proportion of the maximum value.

Persistence Survey Interval	N Carcasses Detected = 5						N Carcasses Detected = 10						Fatality Search Interval
	GenEst		EIA F - LP		EIA F - median		GenEst		EIA F - LP		EIA F - median		
	mean %	sd %	mean %	sd %	mean %	sd %	mean %	sd %	mean %	sd %	mean %	sd %	
Continuous	-3.0	10.4	1.3	6.5	1.3	6.5	-0.1	10.5	1.6	6.6	1.6	6.6	I = 3
Mandatory	-0.6	12.3	7.8	11.3	0.6	6.2	2.4	12.4	8.3	11.2	0.9	6.1	
1 day	-0.8	12.0	3.7	8.3	1.2	6.5	2.3	12.1	4.1	8.5	1.6	6.6	
2 days	-1.0	13.4	6.1	10.1	0.8	6.1	2.7	13.9	6.7	10.7	1.1	6.3	
3 days	-0.9	14.7	8.7	12.5	0.2	5.7	3.8	16.7	9.5	12.9	0.5	5.8	
4 days	2.1	19.2	11.1	15.1	-0.7	5.2	7.3	22.2	12.2	16.3	-0.3	5.3	
5 days	12.1	25.5	15.8	21.2	-1.0	5.0	18.4	29.8	16.5	21.1	-0.7	4.9	
Continuous	0.4	17.6	2.8	13.5	2.8	13.5	1.9	18.1	3.3	13.5	3.3	13.5	I = 7
Mandatory	1.5	17.9	16.4	23.9	1.4	12.6	2.9	17.7	17.3	23.5	1.8	12.4	
1 day	1.1	17.2	7.8	17.3	2.7	13.4	2.7	17.4	8.6	17.5	3.3	13.5	
2 days	0.8	17.4	12.8	21.3	1.8	12.7	2.7	17.8	13.9	22.3	2.4	12.9	
3 days	0.6	17.6	18.4	26.5	0.6	11.8	3.2	18.7	19.8	27.1	1.2	11.8	
4 days	1.9	20.7	23.7	33.1	-1.3	10.6	5.0	23.1	25.7	35.3	-0.6	10.8	
5 days	10.5	28.1	35.3	53.8	-1.8	10.2	14.1	31.2	35.9	54.5	-1.4	9.9	
Continuous	5.2	26.4	5.6	22.4	5.5	22.3	7.4	26.9	5.3	21.9	5.3	21.8	I = 14
Mandatory	4.0	25.1	33.0	57.5	3.1	20.3	5.9	25.1	31.6	52.2	2.9	19.4	
1 day	3.6	24.8	15.0	34.2	5.5	22.5	5.5	24.7	14.3	32.7	5.3	22.0	
2 days	2.9	24.1	25.7	50.3	3.8	20.6	5.2	24.5	25.2	49.6	3.9	20.6	
3 days	2.8	23.8	41.5	83.6	2.1	19.4	5.0	23.6	38.4	68.1	1.9	18.3	
4 days	1.8	23.5	55.8	107.1	-1.0	16.8	4.7	25.2	54.4	100.4	-0.9	16.7	
5 days	7.2	31.7	92.1	203.4	-1.9	15.9	9.9	32.5	82.8	160.9	-2.1	15.3	

estimators provided results close to each other. However, mortality estimates produced by each estimator tended to differ when persistence interval survey lengthened. The GenEst estimator and EIAF-median produced similar mortality estimates, but EIAF-LP significantly differed from the other estimators. Contrary to our expectations, the bias and accuracy of the three tested estimators were marginally affected by the number of carcasses found, a finding contrasting with those of previous studies, which found a decline performance of the approaches tested when number of carcasses detected decreased (Huso et al., 2015; Péron, 2018). Conversely, the estimates produced by GenEst appeared to exhibit a slightly more positive bias when the number of detected carcasses was higher. The minimal variation observed in response to an increase in carcass detected suggests that the estimated bias and precision of the estimators could be extrapolated a dataset with a higher number of carcasses detected.

The findings demonstrate the ability of GenEst to estimate mortality with low bias and high accuracy, even when the time interval between carcass persistence surveys increased. Comparing the results obtained from a realistic sample of 15 carcass persistence observations to a larger dataset of 218 observations shows the robustness of this estimator for estimating mortality by correcting for the persistence probability obtained from a small sample of carcasses, at least as low as that usually achieved in EIAs. Nevertheless, the slight increase in bias and decrease in accuracy observed in this method as the fatality search interval extended to 14 days suggests that it would be preferable to increase the search frequency to 3 or 7 days for obtaining more reliable results. The similarity between the mortality estimates obtained with GenEst using the continuous survey and the resampled mandatory survey intervals used in EIAs suggests that this estimator is able to provide accurate and unbiased estimates from mandatory persistence trials. This may be due to the fact that GenEst allows fitting a survival model using the best distribution family given the data, allowing a robust estimate of persistence even when the frequency and number of persistence checks are lower. However, the higher bias and lower accuracy of the estimates when conducting the carcass survey every 4 or 5 days shows that carcass persistence surveys should be conducted at a frequency equal to or <3 days.

The two estimators based on the formulas used for EIA of windfarms in some countries produced similar mortality estimates when using continuous carcass persistence observation (i.e. observed by camera traps). However, their results strongly diverged when persistence survey intervals lengthened, highlighting the significant impact of the choice of the mean persistence calculation method. Calculating the mean persistence time as the average time between carcass placement and last presence observed during persistence trials (EIAF-LP estimator) resulted in an underestimation of persistence time compared to the other two methods, leading to an overestimation of mortality. Increasing the interval between searches strongly increased bias and decreased the accuracy of mortality estimates. Indeed, using the last presence observed to calculate mean persistence leads to its underestimation, as the removal event has necessarily occurred after the last presence observed, and the wider the interval, the greater the underestimation. Based on these findings, it is likely that using last presence for persistence estimation will lead to overestimation of mortality associated with windfarms when persistence is not observed continuously. Therefore, we suggest this method should not be used to calculate persistence time.

In contrast, estimating mortality using the median time between the last observed presence and first absence of the carcass (EIAF-median) resulted in less biased and more accurate mortality estimates in comparison to the EIAF-LP estimator. The mortality estimates produced with EIAF-median were close to those obtained with GenEst on the same subsample of carcass persistence. Unlike EIAF-LP, increasing intervals between persistence surveys did not strongly impact the bias and accuracy of the estimates. Increasing carcass survey intervals results in less frequent checks over the same period of time; this reduces the number of possible values when calculating the median between the last presence and first absence observed. Increasing persistence survey intervals then leads to an apparent, but artefactual, increase in the accuracy of this estimator. Although the median provides a better approximation of mean persistence time than the last observed presence, given that the probability of a carcass disappearing decreases over time, using the median leads to a slight overestimation of persistence. This overestimation increases with the length of the persistence survey interval. This explains why longer persistence survey intervals lead to lower mortality estimates when using this estimator. Thus, increasing persistence survey intervals will underestimate the mortality of flying fauna when using this estimator.

Comparing the results of the EIAF-median estimator with the GenEst reference value indicated a higher bias than comparing it with its own reference value. These results suggest that the exponential family assumed in the EIAF method, implying a constant removal hazard, may not adequately describe carcass persistence over time for the empirical data collected in the field, as previously highlighted in comparative studies (Bispo et al., 2013a). By allowing several distribution families to be tested to fit the carcass persistence model, the GenEst method better depicts the survival probability of carcasses over time, and ensures this parameter is integrated in the estimation of mortality (Bispo et al., 2013a, 2013b; Dalthorp et al., 2018; Rabie et al., 2021). Another point not directly addressed in this study is the influence of carcass persistence trial duration on the estimation of mean persistence time. The EIAF estimator used for EIAs uses mean persistence time to estimate mortality; however, given that carcasses may lose interest for scavengers after their desiccation, particularly in dry environments (Bispo et al., 2013a; Péron, 2018), a longer trial duration will lead to an increase in the mean persistence estimate in the presence of right-censored data (Zhong and Hess, 2009), which in turn will bias mortality estimates. By using a survival model rather than the estimation of mean persistence time, GenEst may be less biased by the duration of carcass persistence trials.

4.2. Recommendations

In light of these findings, we advocate for restricting the duration between persistence surveys (preferably not exceeding 3 days) and employing the GenEst estimator for mortality estimation.

Wide fatality search intervals increased bias and reduced the accuracy of all the estimators we tested. The impact of fatality search intervals on mortality estimation has been previously highlighted, with results showing that reducing intervals between searches may reduce bias when estimating mortality (Huso, 2011; Korner-Nievergelt et al., 2015; Rabie et al., 2021). We therefore strongly recommend increasing the frequency of fatality searches up to every 3 days in order to reduce the uncertainty of mortality estimates, particularly when carcass persistence is low.

Large differences in bias and accuracy were found between the estimators we tested. The results clearly suggest that using the last presence to estimate the mean persistence time of a carcass resulted in strong overestimation of mortality. The EIAF-LP estimator should thus be avoided, and the median time between the last presence observed of a carcass and its first absence should be preferred when using the EIAF formula (i.e. EIAF-median). Although EIAF-median and GenEst estimators showed similar bias and accuracy in this study, the performance of the former estimator may be an artefact of the median calculation method, as described above. We therefore recommend using the GenEst estimator when estimating mortality. Although the use of this estimator requires some knowledge of the underlying statistic methodology involved—for the model performance selection, for example—the existing documentation makes it easy to use by following a step-by-step procedure (Simonis et al., 2018). In addition, GenEst can be parametrized for other parameters, by including the search schedule, the density-weighted proportion, as well as the effect of carcass size and season, which have been shown to have an impact on persistence probability and mortality rates (Arnett et al., 2008; Bernardino et al., 2022; DeVault et al., 2017; DeVault et al., 2004).

During our carcass persistence trials, 77 % of the carcasses were removed before the end of the survey (14 days), and >90 % of those carcasses were removed during the first 7 days. This suggests that carrying out the persistence trial for 14 days as often recommended for EIAs is a good trade-off between the monitoring effort involved and the accuracy of the results when using the GenEst estimator, which allows a survival curve to be modeled. In contrast, we do not recommend using EIAF-LP or EIAF-median when a high proportion of carcasses are still present at the end of the study, as the mean persistence time would be miscalculated, as highlighted above. The higher probability of removal at the beginning of the persistence trial also points to the fact that monitoring effort should be increased during the first days following carcass placement—at least in France, where the study was carried out. However, this must be determined locally and seasonally, as persistence time can vary widely (e.g. mean persistence time in a tropical environment in Mexico: 2.0 to 4.4 days, Villegas-Patraca et al., 2012; mean persistence time in various sites across Italy: 1.2 days to 25.8 days, Costantini et al., 2017). In cases in which the mean persistence time is shown to be very low, as in the tropics, more frequent persistence checks and fatality searches may be required.

Finally, we also recommend the use of camera traps to assess the persistence time of carcasses, as this allows removal events to be determined with great accuracy, thus reducing the potential bias associated with this parameter. In this study, <7 % of the carcass removals could not be dated with an accuracy of less than an hour. However, depending on the model, batteries often need to be changed to prevent the loss of information, as was the case during our fieldwork. This method allows a significant reduction of field work, and their purchase cost is rapidly amortized (Rosa et al., 2019). Furthermore, the use of camera traps allows the identification of the predator guilds responsible for the removal events (Paula et al., 2014; Rosa et al., 2019), which could be used to further study the biogeographic variation of persistence time associated with the type of predators present.

5. Conclusion

The results of this study allowed easily interpretable and applicable recommendations for the estimation of carcass persistence in the field. Although the focus of the study was the estimation of mortality linked to windfarms, the recommendations could be relevant for other infrastructure responsible for wildlife mortality. Developing a shared and standardized protocol would be highly valuable for estimating wildlife mortality caused by human infrastructure and would facilitate the comparison of countries and biogeographic regions. Moreover, it paves the way for assessing cumulative impacts on migratory species at a continental scale. This would enable a better understanding of how windfarms and other human-made structures affect bird and bat populations so that these adverse effects could be more effectively mitigated.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2024.110509.

CRediT authorship contribution statement

Andreas Ravache: Conceptualization, Data curation, Formal analysis, Methodology, Project administration, Visualization, Writing – original draft, Writing – review & editing. **Kévin Barré:** Conceptualization, Formal analysis, Methodology, Writing – original draft, Writing – review & editing. **Brice Normand:** Conceptualization, Funding acquisition, Methodology, Writing – original draft, Writing – review & editing. **Corentin Goislot:** Conceptualization, Methodology. **Aurélien Besnard:** Conceptualization, Formal analysis, Funding acquisition, Methodology, Writing – original draft, Writing – review & editing. **Christian Kerbiriou:** Conceptualization, Formal analysis, Funding acquisition, Methodology, Writing – original draft, Writing – review & editing.

Declaration of competing interest

At the time of submission one of the authors, Brice Normand, was an employee of Ouest Am', an environmental consultancy involved in wind turbine impact assessment studies. In addition, Kévin Barré was funded by ADEME, a public agency promoting renewable energies. Authors thus declare a direct conflict of interest according to Biological Conservation journal ethics. Authors take complete responsibility for the integrity of the data and the accuracy of their analysis.

Data availability

The data that has been used is confidential.

Acknowledgements

We thank the Ouest Am' team for their help during fieldwork: C. Tanton, L. Hubert, L. Bellion, Q. Membrut, S. Desdoits & T. Perrono. We wish to thank the wind farm operators for allowing access to the wind turbines where the study was conducted. We also thank M. Huso for her feedback on the protocol used in this study and on the interpretation of its correction formula as used by the environmental consultants.

Funding

Funds were provided by the French government to support Andreas Ravache's post-doctoral position. (France Relance plan, n°ANR-21-PRRD-0001-01).

References

APA, 2010. Guia metodológico para a avaliação de impacte ambiental de parques eólicos. Agência Portuguesa do Ambiente.

Arnett, E.B., Brown, W.K., Erickson, W.P., Fiedler, J.K., Hamilton, B.L., Henry, T.H., Jain, A., Johnson, G.D., Kerns, J., Koford, R.R., Nicholson, C.P., O'Connell, T.J., Piorkowski, M.D., Tankersley, R.D., 2008. Patterns of bat fatalities at wind energy facilities in North America. J. Wildl. Manag. 72, 61–78. https://doi.org/10.2193/2007-221.

AWWI, 2020. American Wind Wildlife Institute Technical Report: 2nd Edition: Summary of Bat Fatality Monitoring Data Contained in AWWIC. Washington, DC.

Barré, K., Le Viol, I., Bas, Y., Julliard, R., Kerbiriou, C., 2018. Estimating habitat loss due to wind turbine avoidance by bats: Implications for European siting guidance. Biol. Conserv. 226, 205–214. https://doi.org/10.1016/j.biocon.2018.07.011.

Barrientos, R., Martins, R.C., Ascensão, F., D'Amico, M., Moreira, F., Borda-de-Água, L., 2018. A review of searcher efficiency and carcass persistence in infrastructure-driven mortality assessment studies. Biol. Conserv. 222, 146–153. https://doi.org/10.1016/j.biocon.2018.04.014.

Bernardino, J., Martins, R.C., Bispo, R., Marques, A.T., Mascarenhas, M., Silva, R., Moreira, F., 2022. Ecological and methodological drivers of persistence and detection of bird fatalities at power lines: insights from multi-project monitoring data. Environ. Impact Assess. Rev. 93, 106707 https://doi.org/10.1016/j.eiar.2021.106707.

Bispo, R., Bernardino, J., Marques, T.A., Pestana, D., 2013a. Modeling carcass removal time for avian mortality assessment in wind farms using survival analysis. Environ. Ecol. Stat. 20, 147–165. https://doi.org/10.1007/s10651-012-0212-5.

Bispo, R., Bernardino, J., Marques, T.A., Pestana, D., 2013b. Discrimination between parametric survival models for removal times of bird carcasses in scavenger removal trials at wind turbines sites. In: Stud. Theor. Appl. Stat. Sel. Pap. Stat. Soc, pp. 65–72. https://doi.org/10.1007/978-3-642-34904-1_6.

Carrete, M., Sánchez-Zapata, J.A., Benítez, J.R., Lobón, M., Donázar, J.A., 2009. Large scale risk-assessment of wind-farms on population viability of a globally endangered long-lived raptor. Biol. Conserv. 142, 2954–2961. https://doi.org/10.1016/j.biocon.2009.07.027.

Costantini, D., Gustin, M., Ferrarini, A., Dell'Omo, G., 2017. Estimates of avian collision with power lines and carcass disappearance across differing environments. Anim. Conserv. 20, 173–181. https://doi.org/10.1111/acv.12303.

Dalthorp, D., Madsen, L., Huso, M., Rabie, P., Wolpert, R., Studyvin, J., Simonis, J., Mintz, J., 2018. GenEst statistical models - a generalized estimator of mortality. Autom. Data Process. Comput. 7, 1–13.

DeVault, T.L., Brisbin, I.L., Rhodes, O.E., 2004. Factors influencing the acquisition of rodent carrion by vertebrate scavengers and decomposers. Can. J. Zool. 82, 502–509. https://doi.org/10.1139/z04-022.

DeVault, T.L., Seamans, T.W., Linnell, K.E., Sparks, D.W., Beasley, J.C., 2017. Scavenger removal of bird carcasses at simulated wind turbines: does carcass type matter? Ecosphere 8. https://doi.org/10.1002/ecs2.1994.

Duriez, O., Pilard, P., Saulnier, N., Boudarel, P., Besnard, A., 2023. Windfarm collisions in medium-sized raptors: even increasing populations can suffer strong demographic impacts. Anim. Conserv. 26, 264–275. https://doi.org/10.1111/acv.12818.

Erickson, W.P., Jeffrey, J., Kronner, K., Bay, K., 2004. Stateline Wind Project Wildlife. Monitoring Final Report, July 2001 - December 2003., Technical Report Peer-reviewed by and Submitted to FPL Energy, the Oregon Energy Facility Siting Council, and the Stateline Technical Advisory Committee, Oregon, USA.

Erickson, W.P., Wolfe, M.M., Bay, K.J., Johnson, D.H., Gehring, J.L., 2014. A comprehensive analysis of small-passerine fatalities from collision with turbines at wind energy facilities. PLoS One 9. https://doi.org/10.1371/journal.pone.0107491.

Frick, W.F., Baerwald, E.F., Pollock, J.F., Barclay, R.M.R., Szymanski, J.A., Weller, T.J., Russell, A.L., Loeb, S.C., Medellin, R.A., McGuire, L.P., 2017. Fatalities at wind turbines may threaten population viability of a migratory bat. Biol. Conserv. 209, 172–177. https://doi.org/10.1016/j.biocon.2017.02.023.

Horn, J.W., Arnett, E.B., Kunz, T.H., 2008. Behavioral responses of bats to operating wind turbines. J. Wildl. Manag. 72, 123–132. https://doi.org/10.2193/2006-465.

Huso, M.M.P., 2011. An estimator of wildlife fatality from observed carcasses. Environmetrics 22, 318–329. https://doi.org/10.1002/env.1052.

Huso, M.M.P., Dalthorp, D., Dail, D., Madsen, L., 2015. Estimating wind-turbine-caused bird and bat fatality when zero carcasses are observed. Ecol. Appl. 25, 1213–1225. https://doi.org/10.1890/14-0764.1.

Huso, M., Som, N., Ladd, L., 2018. Fatality Estimator User's Guide: Data Series, p. 729.

IRENA, 2019. Global Energy Transformation: A Roadmap to 2050. Abu Dhabi.

Kitano, M., Ino, M., Smallwood, K.S., Shiraki, S., 2020. Seasonal difference in carcass persistence rates at wind farms with snow, Hokkaido. Japan. Ornithol. Sci. 19, 63–71. https://doi.org/10.2326/osj.19.63.

Korner-Nievergelt, F., Korner-Nievergelt, P., Behr, O., Niermann, I., Brinkmann, R., Hellriegel, B., 2011. A new method to determine bird and bat fatality at wind energy turbines from carcass searches. Wildl. Biol. 17, 350–363. https://doi.org/10.2981/10-121.

Korner-Nievergelt, F., Behr, O., Brinkmann, R., Etterson, M.A., Huso, M.M.P., Dalthorp, D., Korner-Nievergelt, P., Roth, T., Niermann, I., 2015. Mortality estimation from carcass searches using the R-package carcass-a tutorial. Wildl. Biol. 21, 30–43. https://doi.org/10.2981/wlb.00094.

Kumar, Y., Ringenberg, J., Depuru, S.S., Devabhaktuni, V.K., Lee, J.W., Nikolaidis, E., Andersen, B., Afjeh, A., 2016. Wind energy: trends and enabling technologies. Renew. Sust. Energ. Rev. 53, 209–224. https://doi.org/10.1016/j.rser.2015.07.200.

Leroux, C., Kerbiriou, C., Le Viol, I., Valet, N., Barré, K., 2022. Distance to hedgerows drives local repulsion and attraction of wind turbines on bats: implications for spatial siting. J. Appl. Ecol. 1–12 https://doi.org/10.1111/1365-2664.14227.

Leung, K.M., Elashoff, R.M., Afifi, A.A., 1997. Censoring issues in survival analysis. Annu. Rev. Public Health. https://doi.org/10.1146/annurev.publhealth.18.1.83.

Millon, L., Colin, C., Brescia, F., Kerbiriou, C., 2018. Wind turbines impact bat activity, leading to high losses of habitat use in a biodiversity hotspot. Ecol. Eng. 112, 51–54. https://doi.org/10.1016/j.ecoleng.2017.12.024.

MTERD, 2020. Propuesta de Directrices para la Evaluación y Corrección de la Mortalidad de Quirópteros en Parques Eólico. Ministerio para la Transición Ecológica y el Reto Demográfico, Spain.

MTES, 2018. Protocole de suivi environnemental des parcs éoliens terrestres - révision 2018.

Paula, J.J.S., Bispo, R.M.B., Leite, A.H., Pereira, P.G.S., Costa, H.M.R.G., Fonseca, C.M.M.S., Mascarenhas, M.R.T., Bernardino, J.L.V., 2014. Camera-trapping as a methodology to assess the persistence of wildlife carcasses resulting from collisions with human-made structures. Wildl. Res. 41, 717–725. https://doi.org/10.1071/WR14063.

Péron, G., 2018. Process-based vs. ad-hoc methods to estimate mortality using carcass surveys data: a review and a note about evidence complacency. Ecol. Model. 384, 111–118. https://doi.org/10.1016/j.ecolmodel.2018.06.021.

Rabie, P.A., Riser-Espinoza, D., Studyvin, J., Dalthorp, D., Huso, M., 2021. Performance of the GenEst Mortality Estimator Compared to the Huso and Shoenfeld Estimators. American Wind Wildlife Institute, Washington, DC.

Rosa, L., Neves, T., Vieira, D., Mascarenhas, M., 2019. Camera-trapping versus conventional methodology in the assessment of carcass persistence for fatality estimation at wind farms. Wind Energy Wildl. Impacts 165–177. https://doi.org/10.1007/978-3-030-05520-2_11.

Schippers, P., Buij, R., Schotman, A., Verboom, J., van der Jeugd, H., Jongejans, E., 2020. Mortality limits used in wind energy impact assessment underestimate impacts of wind farms on bird populations. Ecol. Evol. 10, 6274–6287. https://doi.org/10.1002/ece3.6360.

Scholz, C., Voigt, C.C., 2022. Diet analysis of bats killed at wind turbines suggests large-scale losses of trophic interactions. Conserv. Sci. Pract. 4, 1–12. https://doi.org/10.1111/csp2.12744.

Simonis, J., Dalthorp, D., Huso, M., Mintz, J., Madsen, L., Rabie, P., Studyvin, J., Huso, M., Mintz, J., Madsen, L., Rabie, P., Studyvin, J., 2018. GenEst User Guide—Software for a Generalized Estimator of Mortality (No. 7-C19). US Geol. Surv., p. 72

Smallwood, K.S., 2007. Estimating wind turbine–caused bird mortality. J. Wildl. Manag. 71, 2781–2791. https://doi.org/10.2193/2007-006.

SNH, 2009. Guidance on Methods for Monitoring Bird Populations at Onshore Wind Farms. Scottish Natural Heritage.

Turkson, A.J., Ayiah-Mensah, F., Nimoh, V., 2021. Handling censoring and censored data in survival analysis: a standalone systematic literature review. Int. J. Math. Math. Sci. 2021 https://doi.org/10.1155/2021/9307475.

Urquhart, B., Hulka, S., Duffy, K., 2015. Game birds do not surrogate for raptors in trials to calibrate observed raptor collision fatalities. Bird Study 62, 552–555. https://doi.org/10.1080/00063657.2015.1053751.

Villegas-Patraca, R., Macías-Sánchez, S., MacGregor-Fors, I., Muñoz-Robles, C., 2012. Scavenger removal: bird and bat carcass persistence in a tropical wind farm. Acta Oecol. 43, 121–125. https://doi.org/10.1016/j.actao.2012.06.004.

Zhong, M., Hess, K.R., 2009. Mean survival time from right censored data. Collect. Biostat. Res. Arch. 66, 99.