

Estimating habitat loss due to wind turbine avoidance by bats: Implications for European siting guidance

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ABSTRACT

Wind energy is rapidly growing as a renewable source of energy but is not neutral for wildlife, especially bats. Whereas most studies have focused on bat mortality through collision, very few have quantified the loss of habitat use resulting from the potential negative impact of wind turbines, and none of them for hub heights higher than 55 m. Such impacts could durably affect populations, creating a need for improvement of knowledge to integrate this concern in implementation strategies. We quantified the impact of wind turbines at different distances on the activity of 11 bat taxa and 2 guilds. We compared bat activity at hedgerows (207 sites) located at a distance of 0–1000 m from wind turbines ($n = 151$) of 29 wind farms in an agricultural region in the autumn (overall 193,980 bat passes) using GLMMs. We found a significant negative effect of proximity to turbines on activity for 3 species (*Barbastella barbastellus*, *Nyctalus leisleri*, *Pipistrellus pipistrellus*), 2 species-groups (*Myotis* spp., *Plecotus* spp.) and 2 guilds (fast-flying and gleaner). Bat activity within 1000 m of wind turbines by gleaners and fast-flying bats is reduced by 53.8% and 19.6%, respectively. Our study highlighted that European recommendations (at least 200 m from any wooded edge) to limit mortality events likely strongly underestimate the loss of bat activity. The current situation is particularly worrying, with 89% of 909 turbines established in a region that does not comply with recommendations, which themselves are far from sufficient to limit the loss of habitat use.

1. Introduction

Land consumption due to the development of projects (e.g., transport infrastructure, power generation infrastructure, and urbanization) is a major driver of biodiversity loss (Maxwell et al., 2016). Project developers should avoid and reduce their negative impacts on biodiversity as much as possible and implement offset measures when residual effects persist (mitigation hierarchy, EC, 2007). Assessment studies before projects are set up aim to quantify impacts (i.e., direct loss of individuals and future habitat losses) in order to apply the mitigation hierarchy. Most of these studies mainly focus on habitat losses; however, wind farms are an exception because of weak covered area in

the construction stage and growing concerns about impacts to wildlife issues in the post-construction stage (Gibson et al., 2017).

A large number of studies summarized by Arnett et al. (2016) have shown that wind farms have adverse effects on bats through mortality events from collisions in the post-construction stage and could threaten population viability (Frick et al., 2017). Whereas many studies have focused on bat mortality through collision with wind turbines, few have studied activity loss in the post-construction stage resulting from the potential impact on habitat use around wind farms. Habitat availability, notably foraging habitat, is nevertheless recognized as a major driver of population dynamics for most taxa (Ney-nifle and Mangel, 2000; Rybicki and Hanski, 2013). This is especially the case for one in-

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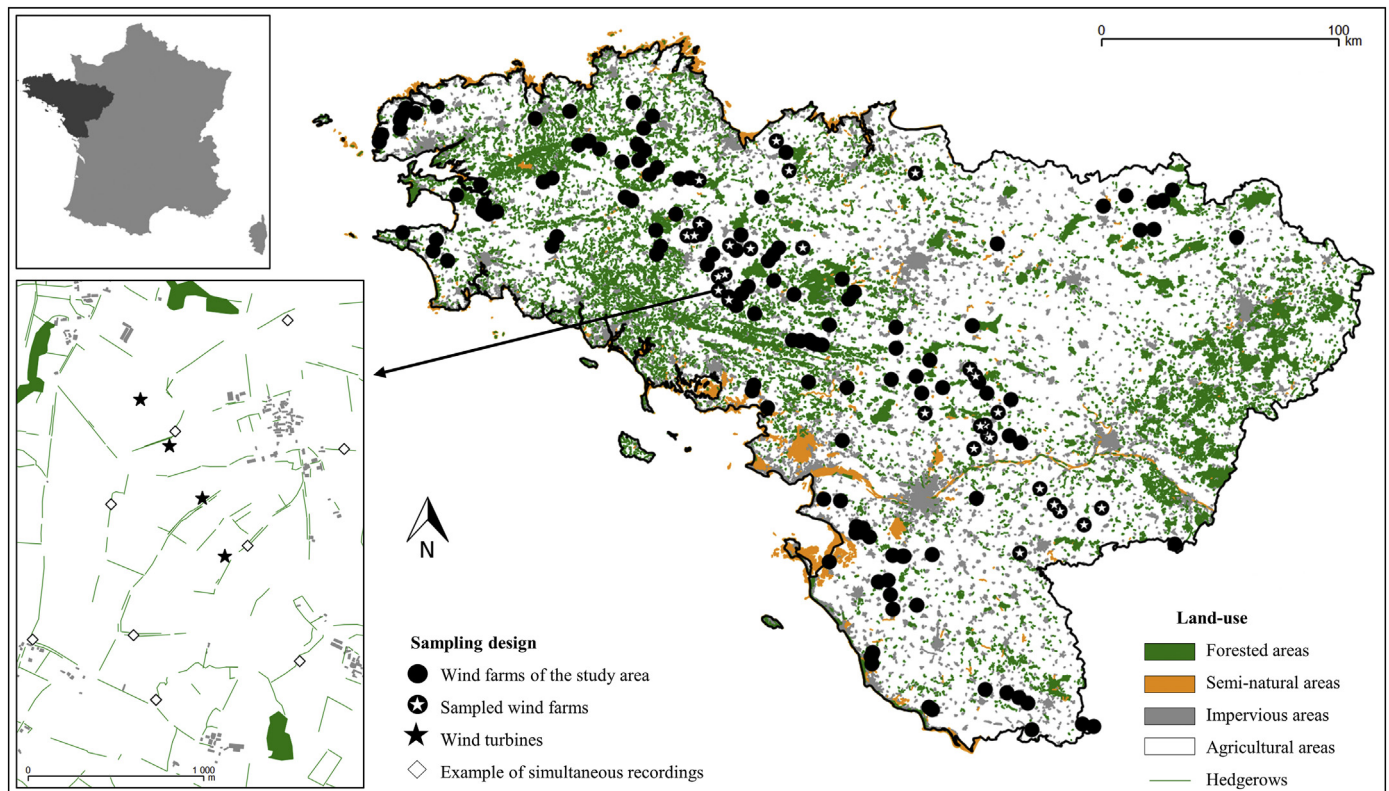


Fig. 1. Map of the land use, total and studied wind turbines in the study region, showing an example of sampling with simultaneous recordings of bat activity over one night.

sectivorous bat species, *Rhinolophus ferrumequinum*, whose colony size strongly depends of the density of hedgerows surrounding the roosts (Froidevaux et al., 2017). The establishment of wind farms, by modifying environmental conditions, may thus durably affect the habitat use of such long-lived species with high survival rates (e.g. 0.80 for *Pipistrellus pipistrellus* and 0.91 for *R. ferrumequinum*; Sendor and Simon, 2003; Schaub et al., 2007) and low fecundity (e.g. 0.72 for *P. pipistrellus* and 0.74 for *R. ferrumequinum*; Webb et al., 1996; Schaub et al., 2007). Moreover, agricultural landscapes are widely used by bats as foraging areas (Boyles et al., 2011; Wickramasinghe et al., 2004). Indeed, some widespread habitats in agricultural areas are known to be essential for bats, such as wetlands (Sirami et al., 2013) and hedgerows (Lacoeuilhe et al., 2016), structuring the landscape used by bats (Boughey et al., 2011a; Frey-Ehrenbold et al., 2013).

To our knowledge, only 2 studies have dealt with the impact of the distance of wind turbines on the attractiveness of foraging habitat, and they studied small turbines (< 25 m hub height). Minderman et al. (2012, 2017) found a significant reduction of activity for *P. pipistrellus* and no effect for *Pipistrellus pygmaeus* (2 species studied) up to 400 m from the turbines (between 6 and 25 m hub height). Two other studies have shown a strong reduction in bat activity with proximity to wind turbines without account for impact distances. First, Millon et al. (2015) showed a significantly lower global bat activity within European intensive agricultural fields under wind turbines of 100 m hub height than in fields 35 km away from any turbine. Then, the same authors showed a significantly lower activity (20 times in mean) at wind

turbine sites (between 50 and 55 hub height) than paired sites 1 km away from any turbine for *Miniopterus* sp. and *Chalinolobus* sp. in an island tropical context (Millon et al., 2018). Thus, concerning the standard turbines (> 55 m hub height), there has been no accurate assessment of the distance and the magnitude of the wind turbine impact on the attractiveness of foraging habitat. In addition, overall very few species have been studied in relation to these questions. Another great issue is the reduction of the mortality risk by setting up wind turbines far from attractive habitats such as wooded edges, including hedgerows (Boughey et al., 2011a; Lacoeuilhe et al., 2016). Hedgerows in agricultural landscapes concentrate most of the activity for the majority of bat species, which becomes very low at > 200 m from hedgerows in open areas (Kelm et al., 2014). Guidelines of the Agreement on the Conservation of Populations of European Bats (UNEP/EUROBATS; Rodrigues et al., 2015) have recommended since 2008 that turbines should not be installed closer than 200 m to any types of wooded edges (forests and hedgerows) due to the high risk of fatalities. However, these recommendations only consider the avoidance of collision and are based on the observation of reduced activity with increased distances to wooded edges. Reduction of activity in habitats close to turbines as well as the threshold distance of this impact are not considered in recommendations.

Moreover, the installed capacity of wind energy has grown as a renewable energy source over the last 10 years by a factor of 6.6 (Global Wind Energy Council, 2016). This strong positive trend is expected to continue. Indeed, the 2015 United Climate Change Conference (COP

21) in Paris signed by 195 countries reinforced the development of renewable energy, in which wind energy occupies an important place. Many wind farms are developed in intensive agricultural areas to avoid urban areas and habitats of conservation concern such as forests.

In this context, there is an urgent need to assess the potential decrease in bat activity close to wind turbines in order to quantify the changes of habitat use and the distance of impact. This possible underestimated impact of wind turbines could constitute an important concern, affecting population dynamics with a loss of habitat availability (Rodrigues et al., 2015). Such an approach of assessing the loss of bat activity due to the establishment of new structures could also be helpful to define ecological equivalences in the context of avoidance and offset measures (Millon et al., 2015). This obviously implies being able to assess species-specific loss of activity according to the distance to wind turbines.

We assessed the impact of wind turbines on the bat activity (8 species, 3 species groups and 2 guilds) in a habitat well-recognized for its importance for the species. We designed a study recording bat activity on hedgerows along a uniform gradient of distance (0–1000 m) from 151 turbines of 29 wind farms. Such a design allowed us to evaluate the current loss of activity according to the distance from turbines that can be attributed to their presence. Finally, we assessed how the European recommendations are applied so far, and we estimated the length of deserted hedgerow by bats due to existing wind turbines that this involves.

2. Methods

2.1. Study area

Data were collected at 29 wind farms for a total of 151 turbines across two regions in northwest France (Fig. 1), representing 16.6% of the 909 installed turbines in these regions. All wind farms were composed of 3 to 11 turbines (mean 5.2) of 84 m (standard deviation 12 m) in hub height. For each farm, all turbines had the same height. The installed wind energy capacity in the studied regions was 12,141 MW,

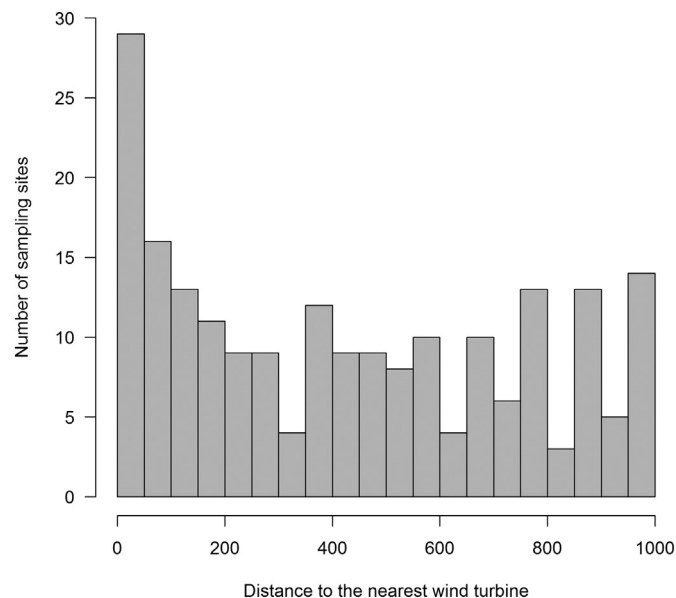


Fig. 2. Number of sampled sites across distances between 0 and 1000 m from the nearest wind turbine.

representing 13.8% of the national installed capacity and covering 5.1% of the electricity needed in these regions (RTE, 2017). The study regions are dominated by agricultural areas (82.2%), whose arable land and grassland compose 48 and 34.2% of the land area, respectively. Urban areas, mainly characterized by villages and small cities, only represent 6.1%, and forests and wetlands cover 10.6 and 1.1%, respectively.

The precise location (geographical coordinates) and establishment date of the studied 909 wind turbines were known thanks to information from the French environmental authority.

2.2. Sampling design and bat data

We studied bat activity at hedgerows along a uniform gradient distance of 0 to 1000 m from the nearest wind turbine (Fig. 2) through recordings of echolocation calls on 207 sites distributed around the 151 wind turbines of the 29 farms studied in operation. To ensure robust comparisons between sites (i.e. different distances from the nearest turbine) of a given wind farm, farms were selected by minimizing the landscape heterogeneity while optimizing the variation in the distance of hedgerows from turbines in the surrounding area. We defined sites for a given wind farm in order to minimize differences in the hedgerow quality (i.e. width and height) and local surrounding habitats (i.e. type of main land-use) and to facilitate accessibility. Only one site per hedgerow was sampled, only once, and sites were separated by at least 300 m from each other. We simultaneously sampled 5–13 sites (average = 9) per night, covering a uniform gradient of available distances from the nearest turbines (Table A.1). Sampling was carried out using bat acoustic records over 23 nights from the 7th of September to the 8th of October 2016 during the migration period (Voigt et al., 2016, 2015). Among the 23 nights, 14 were dedicated to the sampling of only one wind farm per night, while the other 9 nights allowed us to simultaneously sample 2 wind farms per night (these wind farms were on average 8.1 km distant).

Recordings were performed during the entire night, from 30 min before sunset to 30 min after sunrise. Standardized echolocation calls were recorded using one SM2BAT recorder per site. The detectors automatically recorded all ultrasounds that were 6 dB over the background noise, ensuring a large detection range in frequencies. SMX-US microphones were placed at a height of 1.50 m from the ground and oriented upward on a vertical axis.

Since it is impossible to determine the number of individual bats from their echolocation calls, we calculated a bat activity metric (bat passes), calculated as the number of contacts per night per species. Thus, a bat pass was defined as a single or greater echolocation call within a 5-second interval. This interval is considered a good compromise according to bat pass duration among species (Millon et al., 2015). In a first step, echolocation calls were detected and classified to the most accurate taxonomic level, allowing us to assign a confidence index to each bat pass using the software TADARIDA (Bas et al., 2017). In a second step, we performed a manual validation of the automatic identification. A sample of 1811 bat passes of 10 species and 2 groups were randomly double checked manually by KB and YB using the BatSound© software. A mean of 18 (SD = 10) bat passes per class of the confidence index for each species and group were checked, except for *Rhinolophus* species, where all passes were checked due to the low total number (Table A.2). Based on the results of these manual checks, we performed a logistic regression between the success/failure of automatic species assignment (binomial response variable) and the confidence index of the automatic identification (explanatory variable) for

Table 1

Metrics (mean, standard deviation, minimum and maximum) of the raw (not scaled) environmental covariates used in the modelling procedure, according to the buffer size considered for proportion variables.

Landscape variables	Nearest element	Buffer (250 m)	Buffer (500 m)	Buffer (750 m)	Buffer (1000 m)	Range (min-max)
Land cover						
Length of hedgerows (m)	/	1032.0 ± 523.6	3619.0 ± 1641.5	7797.0 ± 3158.3	13,750.0 ± 5135.7	84.3–29,798.0
Arable land (%)	/	52.9 ± 28.0	50.4 ± 20.8	47.7 ± 16.8	46.5 ± 14.8	0.0–99.1
Grass land (%)	/	34.6 ± 27.9	33.8 ± 20.9	33.9 ± 18.1	33.3 ± 16.5	0.0–97.6
Forest (%)	/	1.3 ± 3.8	2.1 ± 4.1	2.9 ± 4.5	3.3 ± 4.6	0.0–26.1
Distances						
Dist. to wind turbine (m)	436.4 ± 318.1	/	/	/	/	8.1–1000.0
Dist. to forest (m)	737.2 ± 520.5	/	/	/	/	53.7–2700.0
Dist. to urban (m)	302.3 ± 165.8	/	/	/	/	6.7–960.4
Dist. to wetland (m)	548.2 ± 367.1	/	/	/	/	1.6–1644.0

each species or group. This allowed us to predict the needed confidence index from the automatic identification process to tolerate a given maximum error risk (Fig. A.1). Then, we filtered the bat dataset on 5 confidence index thresholds corresponding to a predicted maximum error risk between 0.5 and 0.1 (Table A.3) in order to perform analyses on different thresholds of maximum error risk tolerance and check the consistency of the results.

Three groups (*Pipistrellus kuhlii/nathusi*, *Plecotus* spp. and *Myotis* spp.) were constructed because species within these groups were difficult to distinguish from each other, except one species of *Myotis* spp., *Myotis nattereri*, for which echolocation calls are most often characteristic (Siemers and Schnitzler, 2000; Obrist et al., 2004; Barataud, 2015). We also constructed 2 functional groups, the fast-flying species guild, containing *Barbastellus*, *Pipistrellus*, *Eptesicus* and *Nyctalus* genera known to fly along wooded edges and in nearby open environments, and the gleaner species guild, containing *Plecotus* and *Rhinolophus* genera, as well as *Myotis nattereri*, known to fly in cluttered environments (Schnitzler and Kalko, 2001). We did not include *Myotis* spp. in the gleaner species guild due to the diversity of flight behaviours (not always in cluttered environments) of the remaining undetermined species (Schaub and Schnitzler, 2007; Schnitzler and Kalko, 2001).

2.3. Environmental covariates

We selected 7 environmental covariates (the distance to wetlands, forests and urban areas; proportion of arable land, grassland and forest; and length of hedgerows) known as good predictors of bat activity for the species studied (Boughy et al., 2011b; Lacoëuilhe et al., 2016) and which showed enough variability (Table 1). The proportion or length of these environmental covariates was calculated within a 250, 500, 750 and 1000 m radius around the sampling sites in order to use the best scale when selecting those with the smallest Akaike Information Criteria (AIC). Recent landscape data (2016) were provided by the National Institute of Geography (from BD TOPO for data on forests and urban areas, from BD Carthage for wetland data, from BD ORTHO for manual digitization of hedgerows and from Graphical Parcel Register for arable and grass land data (www.ign.fr); distances, lengths and proportions were calculated using ArcGIS 10.0).

The precise location of wind turbines also allowed us to calculate the current distance to the nearest wooded edge (forest or hedgerow) for each turbine in order to describe the current situation with respect to 2008 EUROBATS recommendations.

2.4. Statistical analysis

We assessed whether bat activity of species or groups recorded at hedgerows differed according to the distance to the nearest wind turbine using a Generalized Linear Mixed Model (GLMM, R package *lme4*). According to the nature of the response variable (i.e., number of bat passes) and potential over-dispersion, we chose the best error distribution among Poisson or negative binomial distributions (Zuur et al., 2009).

We included the distance to the nearest wind turbine and the 7 environmental covariates in the models as fixed effects. All variables used in the models were scaled to allow direct comparisons (Schielzeth, 2010). We included interactions between the distance to the nearest wind turbine variable and land cover variables (arable land, grass land and forest proportions, and length of hedgerows) in order to assess the landscape dependence of the wind turbine effects. According to the sampling design (i.e., simultaneous recordings of bat activity along a continuous distance to the nearest wind turbine the same night), we included the date in the models as a factor random variable to control for inter-night variations (e.g., landscape context, weather conditions). Since only one wind farm was sampled per date, rarely two when they were very close, it was not possible to perform models containing, as fixed effects, the farm characteristics, such as height and number of turbines, which were confounded in the random effect. In addition, these characteristics had a low variability (see study area section). For the two separate models on “fast-flying” and “gleaner” species groups, the response variable was the number of bat passes, and the species identity was included as a random effect in order to take into account variation in abundance among species.

Models were fitted on data selected at confidence indices corresponding to a 0.5 maximum error risk tolerance under which data were discarded. This allowed us to conserve a maximum number of bat passes and species occurrences in models (Table A.3). The results were also confirmed at the higher restrictive threshold of confidence indices minimizing the maximum error risk tolerance (0.1) for a majority of species for which data at such a threshold were sufficient (number of bat passes and occurrences).

Full models were constructed by checking correlations between environmental covariates and the distance to the nearest wind turbine and between environmental covariates (Table B.1). We detected a correlation between arable land and grassland covariates ($r > 0.7$); therefore, they were not simultaneously included in the modelling

procedure using the *dredge* function (R package *MuMIn*). The potential non-linear effect of the distance to the nearest wind turbine was checked by visual inspection of the plot from Generalized Additive Mixed Models (GAMM, R package *mgcv*). We detected a quadratic relationship for *N. leisleri* and *Nyctalus noctula* (Fig. B.1); we therefore took this into account in GLMMs for these species by adding a quadratic effect for the distance to wind turbine variable.

We checked that no multicollinearity problems occurred by calculating variance-inflation factors (VIF) using the *corvif* function (R package *AED*; Zuur et al., 2010) on each full model. All variables showed a VIF value < 2, meaning there was no striking evidence of multicollinearity (Chatterjee and Hadi, 2006). Then, we generated from full models a set of candidate models containing all possible variable combinations ranked by corrected AIC (AICc) using the *dredge* function. For each set of candidate models, we performed multi-model inference averaging on a delta AICc < 2 using the *model.avg* function to obtain an averaged regression coefficient for each fixed effect (R package *MuMIn*, Barton, 2015). We used the *allEffects* function (R package *effects*) to get the predicted number of bat passes shown in Fig. 3. From these predictions, we calculated the percentage of lost bat passes as a percentage of the maximum predicted activity for a given species/group. The relative importance of variables as well as the number of candidate models selecting each variable were extracted from the *dredge* procedure. We also checked the non-spatial autocorrelation of residuals of each best model using the *dnearneigh* and *sp.correlogram* functions associated with Moran's I method by visual inspection and significance tests (R package *spatial*, Moran, 1950). We did not detect any problem from the over-dispersion ratio on full and best models (< 1.28; Table B.2). Models were validated by visual examination of residual plots. Since some species had low occurrences, which may cause statistical problems, we confirmed the results of the distance to wind turbine variable by examining models with and without covariates. All analyses were performed using a significance threshold of 5% in the R statistical software v.3.3.1. (R Core Team, 2018).

2.5. Applied quantification of the loss of hedgerow use by bats

To make our results easily transferable to stakeholders (i.e., for loss quantification and offset sizing), we proposed a representation of the impacts on activity. We converted the cumulative loss of bat activity at the landscape scale around wind turbines (1 km) to an equivalent of linear length of deserted hedgerows.

The method presented as an example for a given wind farm in Fig. 4 consists of three main phases:

- 1) The total length of hedgerows (ΣH) in a 1000 m radius was calculated (step 1; Fig. 4).
- 2) With the aim of accounting for the network of hedgerows (length and distance) in the surroundings of the wind farm, each hedgerow in a 1000 m radius was segmented by 10 m sections (steps 2; Fig. 4). The distance of each central point of the hedgerow sections to the nearest turbine was calculated (steps 3; Fig. 4). Then, we calculated the average distance of all sections. This measure corresponds to the averaged distance D of all hedgerows to wind turbines 1000 m around the farm (step 4; Fig. 4).
- 3) Using model predictions (see statistical analysis section), we estimated the corresponding percentage of lost bat activity at this average distance D (%pred_D). This loss is expressed as the percentage of the maximum predicted activity (i.e., activity at 1000 m in our study, see Table B.7). Finally, this loss (%pred_D) was multiplied by the total length of hedgerows (H) to get the length of deserted hedgerows by bats (step 5; Fig. 4).

3. Results

3.1. Bat monitoring

In total, considering a maximum error risk tolerance of 0.5 in the data, 193,980 bat passes of 8 species and 3 species groups were recorded at the 207 study sites, where the most abundant species was *P. pipistrellus*, representing 81% of the observations. The least abundant species were *R. ferrumequinum* (22 bat passes) and *N. noctula* (25 bat passes), which were present in 7 and 9% of the study sites, respectively. All other species or groups were present in > 14% of the study sites (Table 2).

3.2. Impact of wind turbines on bat activity

We detected a significant positive effect of the increasing distance from the nearest wind turbine on the activity of *B. barbastellus*, *Myotis* spp., *N. leisleri*, *P. pipistrellus*, *Plecotus* spp., and fast-flying and gleaner species guilds, plus a significant quadratic effect for *N. leisleri* and a nearly significant quadratic effect for *N. noctula* (Table 3; Fig. 3). This means that the closer a hedgerow was to a wind turbine, the lower was the activity of these species. In contrast, we did not detect any effects for some common (*Eptesicus serotinus*, *Myotis nattereri* and *Pipistrellus kuhlii/nathusii* group) and rare species (*Rhinolophus hipposideros* and *R. ferrumequinum*) (Table 3). There were significant positive interactions of the distance to the nearest wind turbine variable with the proportion of forests for *Myotis* spp., as well as the length of hedgerows for fast-flying and gleaner species guilds. This result suggested that *Myotis* spp. were mainly recorded in contexts of high forest proportion, in which the negative effect of the distance to the nearest wind turbine was significantly higher than in contexts of lower forest proportion. Similarly, for fast-flying and gleaner species guilds, when the hedgerow length was shorter, the activity was higher far from wind turbines. All top candidate models (delta AICc < 2) showed a lower value of AICc than null models (Table B.2). Among all candidate models, the distance to wind turbine variable was always selected, confirming the relative importance of this variable compared to other environmental covariates, except for *E. serotinus* and *Rhinolophus* species (Table B.3).

We also evaluated the potential attenuation of the linear effect of the wind turbine distance by testing significance of a quadratic effect. We did not find a quadratic effect of distance to the nearest wind turbine on activity, except for *N. leisleri* (Fig. 3). Thus, for most species, the negative effect of wind turbines on activity extends at least 1000 m from a wind turbine. The lost activity was therefore likely underestimated and occurred at > 1000 m. Thus, the percentage of lost activity was high, even at long-distances: for instance, at 500 m from the nearest turbine, we detected activity losses of 57% and 77% for *P. pipistrellus* and the gleaner species guild, respectively (Fig. 3).

These results were robust no matter the level of uncertainty included in the identifications (i.e., the maximum error risk tolerance). Indeed, we re-ran the analysis using the most restrictive tolerance of maximum error risk in the data selection of the response variable (0.1), and we found mostly no change in the results for most of the species/groups and guilds (Table B.4). We also compared estimates of models with and without covariates (Table B.5). These negative effects of wind turbines on habitat attractiveness appeared to be little influenced by associated environmental covariates since we did not find any changes for most of the species/groups and guilds when covariates were excluded. Only one change was found for *E. serotinus*, for which the estimate became higher and the *p*-value significant.

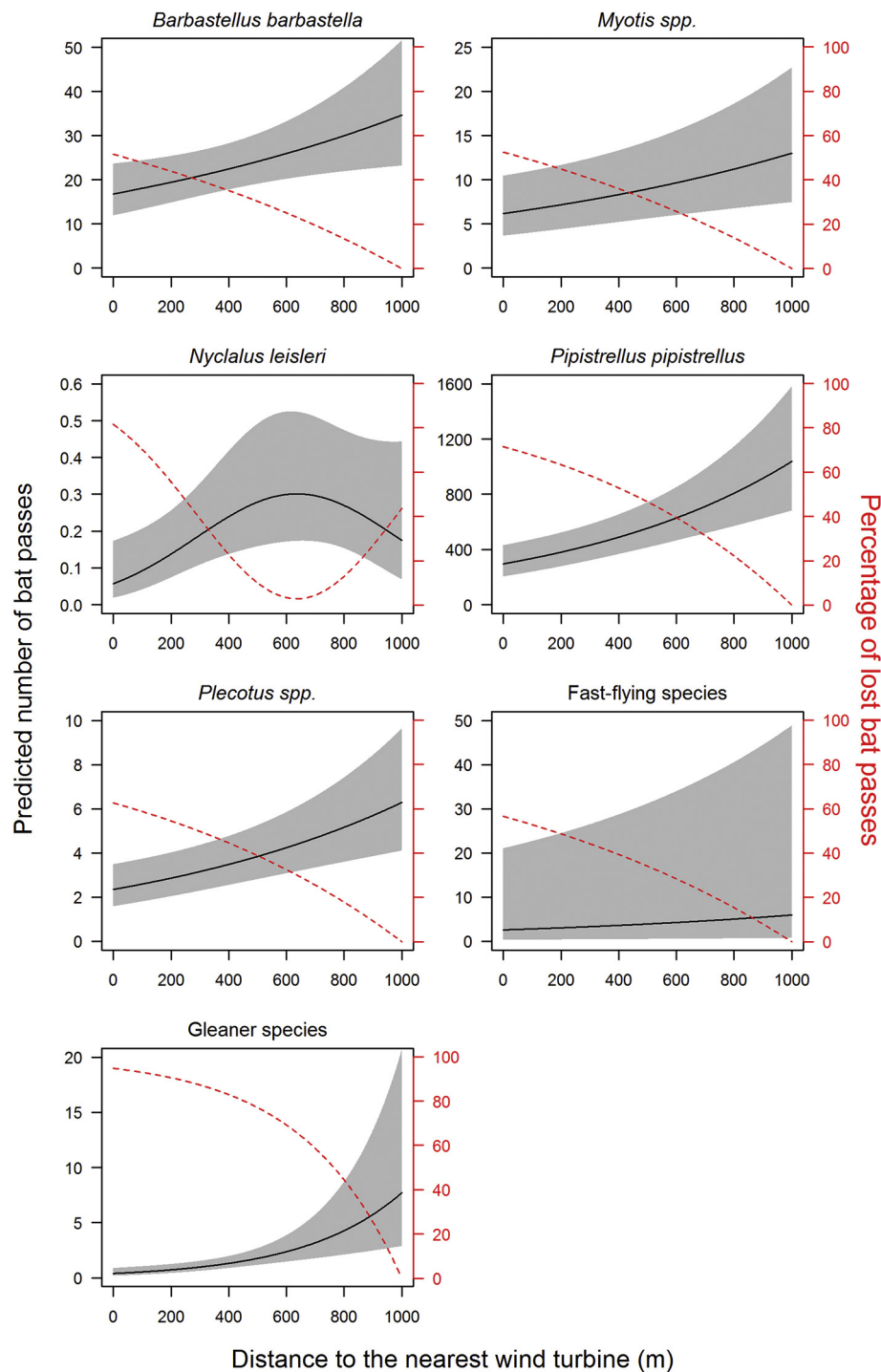


Fig. 3. Variation in the predicted number of bat passes and 95% confidence intervals as a function of the distances to the nearest wind turbine for species/groups and guilds significantly impacted (black continuous curves). Dotted red curves show the corresponding percentage of lost activity calculated from the maximum predicted bat activity. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

3.3. Current wind turbine establishment and generated loss of hedgerow use by bats

Among the 909 wind turbines in northwest France, which contained the studied farms, 89% were established at < 200 m from any type of wooded edges (forest or hedgerows). The situation was the same after the publication of the EUROBATs recommendations in 2008, which recommended nevertheless the avoidance distance of 200 m from any type of edges for the establishment of wind turbines (Fig. 5).

From these current establishments and with our impact assessment method (Fig. 4), making previous statistical results practical for stakeholders, we could quantify the length of deserted hedgerows by bats. Indeed, focusing on the 151 sampled wind turbines, the loss of activity in a 1000 m radius around turbines for fast-flying and gleaner species guilds was 19.6% and 53.8%, respectively, corresponding to 145 and 397 km lengths of deserted hedgerows, respectively. Our 151 studied turbines represent 16.6% of the total number (909) located in northwest France. Thus, by extrapolation, the total length of deserted

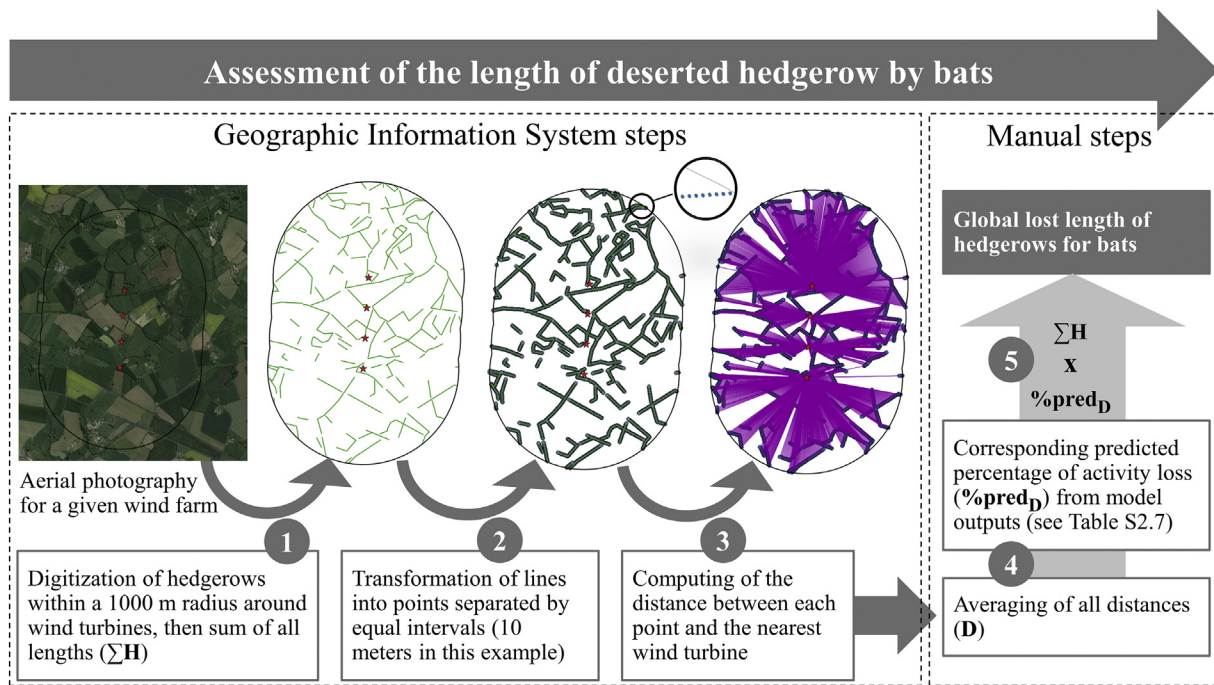


Fig. 4. Steps to assess the length of deserted hedgerow by bats around a given wind turbine/farm using model predictions. QGIS software was used for the digitization, creation of points and calculation of distances.

Table 2

Number of bat passes per species/groups and the corresponding percentage of total passes and occurrences (percentage of presence sites among the 207) according to the applied maximum error risk tolerance for data selection (raw data, maximum error risk tolerance of 0.5 and 0.1).

Species	Number of passes recorded			% of total passes			Occurrence (%)		
	Raw	0.5	0.1	Raw	0.5	0.1	Raw	0.5	0.1
<i>Pipistrellus pipistrellus</i>	159,386	159,386	159,385	81	81	84	99	99	99
<i>Pipistrellus kuhlii/nathusii</i>	24,023	23,603	22,122	12	12	12	98	98	97
<i>Barbastella barbastellus</i>	5479	5472	5436	3	3	3	90	90	90
<i>Myotis</i> spp.	5736	3802	1946	3	2	1	90	86	68
<i>Plecotus</i> spp.	1092	982	566	1	1	< 1	73	72	63
<i>Myotis nattereri</i>	1532	974	439	< 1	< 1	< 1	80	67	45
<i>Eptesicus serotinus</i>	579	543	473	< 1	< 1	< 1	47	42	38
<i>Rhinolophus hipposideros</i>	125	114	110	< 1	< 1	< 1	16	16	15
<i>Nyctalus leisleri</i>	127	53	4	< 1	< 1	< 1	27	16	2
<i>Rhinolophus ferrumequinum</i>	22	22	22	< 1	< 1	< 1	7	7	7
<i>Nyctalus noctula</i>	346	25	8	< 1	< 1	< 1	29	9	3

hedgerows by bats at this scale would be 872 and 2390 km for fast-flying and gleaner species guilds, respectively.

4. Discussion

To our knowledge, this study is the first test of the impact distance of tall wind turbines on bat activity. The results highlight a strong negative effect of turbines on activity, occurring even at distances at least 1000 m for most species, groups and guilds. The negative effect at long distances concerns various species, either fast-flying or gleaner species.

The detected effects are consistent with the few studies dealing with this influence of wind turbines on bat activity (Millon et al., 2018, 2015; Minderman et al., 2017, 2012) and complement them for less intensive agricultural landscapes with high proportions of grasslands and forests that are favourable for bats. Moreover, such a preserved farming landscape constitutes a concern for the conservation of some rare species (e.g., *B. barbastellus* and *Rhinolophus* species) listed in Annex II of the Habitats Directive 92/43/EEC. The undetected

interaction effects of distance with the arable and grassland proportions for all impacted species despite high land-use variability in our data suggest that this negative effect occurs in any agricultural landscape, no matter its composition. However, this result should be confirmed in other more intensive agricultural landscapes.

Negative effects were detected on a wide range of species with highly contrasting ecology and flight behaviour. Some of these species have so far been poorly taken into consideration in environmental studies for wind farm establishment due to a low collision risk (Roemer et al., 2017), thus reinforcing the significance of our findings. Indeed, *B. barbastellus*, *Plecotus* spp. and *Myotis* spp. had a very low level of collision susceptibility index with turbines, taking into account the species abundance (Roemer et al., 2017). However, we did not detect any relationship between the distance to the nearest wind turbine and the *Pipistrellus kuhlii/nathusii* group, which is comprised of a migrant resident species and a long-range migrant species, with an estimate close to zero and small standard error. Flight in migration could be expected to be different, faster and less connected to habitats, compared to flight

Table 3

Estimates and standard errors of the distance to the nearest wind turbine variable (linear and quadratic effects) for the 8 species, 3 species-groups and the 2 guilds studied (** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$, . $P < 0.1$). Complete results of other covariates can be found in Table B.6.

Species	Effect of the distance to the nearest wind turbine on bat activity	
	Linear	Quadratic
<i>Pipistrellus pipistrellus</i>	$0.413 \pm 0.100^{***}$	/
<i>Pipistrellus kuhlii</i> / <i>nathusii</i>	-0.004 ± 0.100	/
<i>Barbastella barbastellus</i>	$0.237 \pm 0.107^*$	/
<i>Myotis</i> spp.	$0.260 \pm 0.091^{**}$	/
<i>Plecotus</i> spp.	$0.309 \pm 0.096^{**}$	/
<i>Myotis nattereri</i>	0.132 ± 0.106	/
<i>Eptesicus serotinus</i>	0.132 ± 0.169	/
<i>Rhinolophus</i> <i>hipposideros</i>	0.099 ± 0.223	/
<i>Nyctalus leisleri</i>	$0.537 \pm 0.208^*$	$-0.413 \pm 0.198^*$
<i>Rhinolophus</i> <i>ferrumequinum</i>	0.329 ± 0.293	/
<i>Nyctalus noctula</i>	0.308 ± 0.290	-0.575 ± 0.307
Fast-flying species	$0.344 \pm 0.123^{**}$	/
Gleaner species	$0.335 \pm 0.068^{***}$	/

closer to the landscape elements in other periods. Therefore, we hypothesize that this absence of effect could be linked to the coexistence of two behaviours in the group, a negative impact on *P. kuhlii* (i.e., avoidance) and a positive impact (i.e., attractiveness) for *P. nathusii*. In this way, migratory species such as *P. nathusii* should be studied at the species level by studying areas in which *P. kuhlii* is absent, as in some areas in Northern Europe (Ancillotto et al., 2016). Finally, we detected a negative effect of the distance to the nearest wind turbine on for *E. serotinus* activity only when we removed covariates. Indeed, this effect was masked by the distance to small urban areas covariate. Even if

there were no striking correlations between these variables and no $VIF > 2$, collinearity issues can occur in some cases when signals are weak and datasets small (Zuur et al., 2010). Thus, to study the effect of wind turbines on *E. serotinus*, it would be more appropriate to choose a study site where colonies locations are known or a study site away from small urban areas.

Mechanisms leading to bats avoidance of environments are still largely unknown and need to be evaluated; one of them could be the avoidance of red aviation lights of wind turbines as suggested by Bennett and Hale (2014) who found a lower number of carcasses under lit compared to unlit turbines. Insectivorous bats may also avoid foraging in noisy environments (Schaub et al., 2009). The negative effects of wind turbines on bat activity that we detected in our study could result from the avoidance of wind turbines surroundings due to these mechanisms. All the wind turbines we studied where lit hence wind turbines lighting could constitute a plausible cause as emitted by Bennett and Hale (2014), even though this was only tested on the carcass number under turbines. Although Schaub et al. (2009) did not include wind turbine noise in their experiment, this could constitute a promising mechanism to test in further studies, in particular for species listening for prey (i.e., passive listening) to find food, which can be highly affected by anthropogenic noise.

4.1. Implications for wind energy development

Despite new recommendations in 2008 from EUROBATS, we showed that the recommendation of a minimal distance of 200 m from woody edges for installing a wind turbine is still far from being considered in most cases. Hence, based on our findings, we first encourage efforts to improve the first step in the application of the mitigation hierarchy proposed by the Business and Biodiversity Offsets Programme, consisting of avoidance by installing turbines as far as possible from edges. Ideally, the EUROBATS recommendation of 200 m is not sufficient. This implies concentrating wind energy in less sensitive

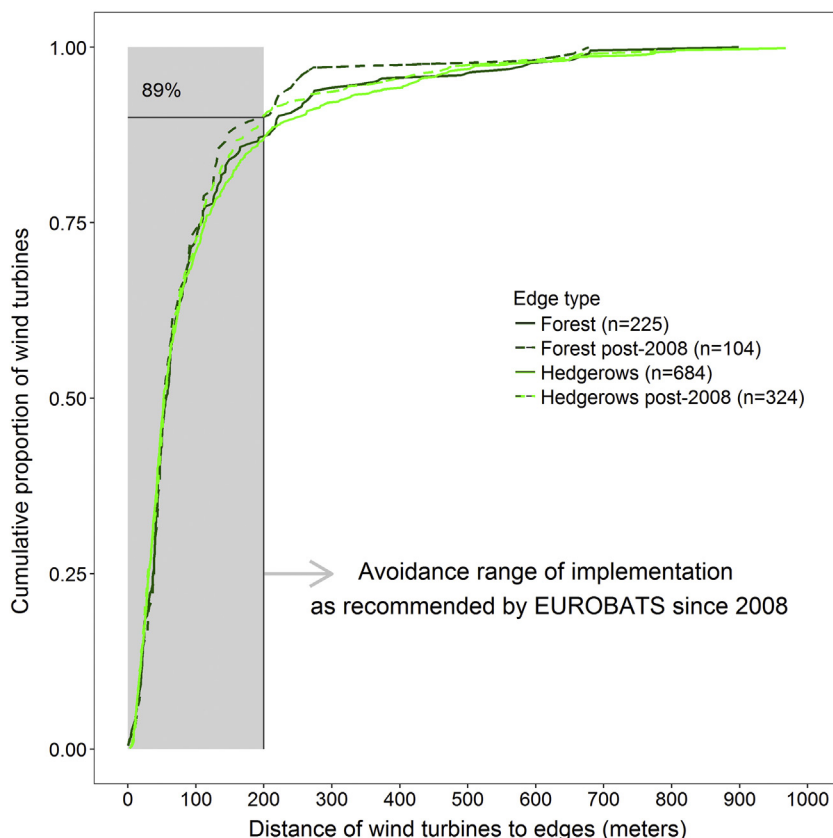


Fig. 5. Distances of the 909 established wind turbines in the study region to the nearest wooded edge (forest or hedgerow), overall and for the post-2008 recommendations period. The grey rectangle under the cumulative curves shows the distance range from the nearest wooded edge which should be avoided in turbine installation according to the EUROBATS recommendations.

areas for bats, far from wooded edges, at > 1000 m. However, such areas could be rare in wooded agricultural regions, thus reflections about wind energy planning and wind farm establishment should be conducted at a large scale to avoid impacts on bats. This is all the more an important concern given the place of bats in the challenge of conciliating agricultural and energy production. Indeed, bats should be considered in this trade-off, considering their huge ecosystem services in agriculture (Boyles et al., 2011; Maine and Boyles, 2015). In addition, for bat conservation, the mitigation of such negative impacts is highly recommended, due to the strong links we hypothesize existing between activity (notably foraging activity) indicative of habitat quality and population dynamics. Indeed, the higher the hedgerow length in the surrounding landscape, the higher the activity (Millon et al., 2015) and the colony size (Froidevaux et al., 2017). In that case, replanting hedgerows on a length that corresponds to the loss of activity could be a helpful measure (Millon et al., 2015). In addition, such offsets require a calculation of the length of deserted hedgerows by bats for sizing, for which our impact assessment method explained in Fig. 4 can be used, making the results transferable to stakeholders. Such an assessment method aims to improve the positioning of turbines in relation to wooded edges and the quantification of offset hedgerows needed, and it can be easily applied by wind turbine operators. However, this method of estimating the length of deserted hedgerows by bats presents some limitations: It is not a physical loss of hedgerow by removal. This means the hedgerow remains potentially functional for a proportion of bat individuals and for other taxa, such as arthropods. The method to assess the loss of an equivalent linear amount of deserted hedgerows is thus an extrapolation to all hedgerows around a given wind farm and only considers bats in the calculation. In addition, the loss metric used assumes that the avoidance of hedgerows by bats during the migration period is also representative of the avoidance of hedgerows by bats that may be resident during other period. Bats use the landscape differently during migration compared to non-migration periods (Millon et al., 2015), hence more data are necessary to get a full picture of the loss of hedgerow use over the course of a year. Moreover, this metric only focuses on hedgerows, which are of high importance for bat populations (Froidevaux et al., 2017), but the use of other types of habitats by bats could also be affected by wind turbines. Finally, the estimated loss is a metric based on models that explained < 30% of the variation in activity in most cases, thus future works should incorporate information about the uncertainty in the model predictions in the loss metric. Our study should encourage operators to stop the installation of wind turbines close to wooded edges, and without offsetting when closer than 1000 m to edges, by objectifying losses and the corresponding need for offsetting. However, in no case should any hedgerows be removed with the aim of reducing impacts on bat activity. The absence of offsetting so far has led to a length of up to 2400 km of deserted hedgerows by bats at the study regions' scale.

Authors' contributions

KB and CK conceived the ideas, KB designed the methodology and collected the data; KB, CK and YB analysed the data; all authors led the writing of the manuscript. All authors critically contributed to the drafts and gave their final approval for publication.

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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.2018.07.011>.

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