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Landscape-scale effects of single- and multiple small wind turbines on bat activity

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Abstract

While the effects of wind farms on bats are widely studied, effects of small wind turbines (SWTs, here <25 m hub height) remain understudied. SWTs are installed in a wider range of habitats compared to wind farms and their effect on wildlife can therefore be different. While single SWTs can adversely affect bat activity in their immediate vicinity, they are often installed in clusters, and to date, no data are available on whether installation of more than one turbine increases the likelihood of adverse effects on bats. Here, we test whether (1) SWT installations affect the activity of two species of bat (the common pipistrelle *Pipistrellus pipistrellus* and soprano pipistrelle *P. pygmaeus*) on a landscape scale (here defined as up to 500 m from SWTs) and (2) such an effect is stronger when multiple (2–4) SWTs are installed. We show that, after accounting for potentially confounding effects (e.g. variation in habitat and weather), (1) mean *P. pipistrellus* activity is lower at 0–100 m compared to 200–500 m from SWTs and (2) the effect on *P. pygmaeus* activity tends to be similar and stronger in multiple SWT sites, although evidence for the latter is limited. We conclude that in some cases, adverse effects of SWTs on bat activity may be measurable over longer spatial scales (within 100 m) than previously thought. However, combined with earlier findings, it is likely that the bulk of such effects operate within relative close proximity of SWTs (<25 m). Moreover, although these effects may be species-specific, with, for example, *P. pygmaeus* potentially more strongly affected by multiple SWT sites, this requires further data. These findings are highly relevant to decision-making aimed at minimizing any adverse effects of wind turbines, specifically single- versus multiple SWT developments, on wildlife.

Introduction

Understanding and predicting how wildlife is affected by disturbance or habitat loss is becoming increasingly important as anthropogenic pressures on the natural world continue to grow (Tuomainen & Candolin, 2011). This is particularly pertinent where incremental changes or disturbances could combine to produce disproportionally greater (adverse) effects. For example, while the rapid growth of the wind energy sector worldwide provides a vital contribution to sustainable energy production, a growing number of studies show that wind farms can have adverse effects on wildlife (Kuvlesky *et al.*, 2007; Zwart *et al.*, 2015), both directly through collision mortality (e.g. Rydell *et al.*, 2010) and indirectly through disturbance or displacement effects (e.g. Pearce-Higgins *et al.*, 2009). Moreover, wind farms with larger numbers of turbines installed can have greater effects on wildlife compared to smaller sites, even if, for example, collision risk per turbine is relatively low (Zwart *et al.*, 2015).

By contrast, similar effects of small wind turbines (SWTs) on wildlife remain relatively unknown. These turbines are not only much smaller in size (in the UK, typical units are between 6 and 25 m hub height) than those in wind farms, they are also commonly installed in a much wider range of habitats. Thus, their impacts on wildlife are likely to be different to that of large turbines, but until recently, this had not been quantified systematically. In addition to direct mortality (Minderman *et al.*, 2014), our previous work found evidence for adverse effects of SWTs on bat activity (e.g. avoidance behaviour, Minderman *et al.*, 2012). While this is likely to lower the risk of collision, such avoidance behaviour could have important population-level consequences in itself, as it can amount to effective habitat loss (Minderman *et al.*, 2012). However, these previous studies focused on a relatively limited spatial scale (e.g. within 25 m of installed

SWTs) and on single SWTs only. Crucially, it is therefore unclear whether any such adverse effects of SWTs on wildlife could be greater at sites with more than one turbine installed, that is, whether the installation of multiple SWTs causes a disproportionally greater effect.

This lack of understanding of the potential disproportionate effects of sites with multiple SWTs is particularly important for two reasons. First, the number of SWTs installed is growing rapidly with over 900 000 units installed worldwide by 2014 (compared to 656 000 in 2010) and this growth is set to continue (Pitteloud and Gsänger 2016). Second, SWTs are increasingly being installed in groups, for example, between two and four turbines in close proximity. Indeed, some companies specifically promote the installation of multiple SWTs to maximize yield and eligibility for government grants or payback schemes. As a result, where SWT installation requires planning permission (Park, Turner & Minderman, 2013), decision makers are increasingly asked to decide whether multiple turbine installations would have greater effects on wildlife, or if limiting the number of SWTs to be installed would be an effective mitigation option. Currently, the evidence base for such decisions for SWTs specifically is entirely lacking.

Here, we aim to (1) quantify the effect of SWTs on bat activity (and thus concomitant habitat loss) on a wider landscape scale, up to 500 m from installed turbines (as opposed to 25 m in Minderman *et al.*, 2014) and (2) to test whether this effect is different between single- and multiple (2–4) turbine installations. Using data collected at 34 SWT sites throughout the UK, we tested the following predictions: (1) bat activity is systematically lower in closer proximity (e.g. 0–100 m from SWTs compared to 100–200 m, 200–300 m, etc.) of operating SWTs, controlling for the effects of habitat and environmental conditions; and (2) the effect of SWT proximity on bat activity is stronger in sites with multiple (2–4) SWTs installed compared to single SWT sites.

Materials and methods

Sites

Data were collected at 34 SWT sites in central and eastern Scotland. Sites were selected from an existing database of owners (Minderman *et al.*, 2012; Park *et al.*, 2013) based on how representative they were of common UK SWT installation settings in terms of habitat, turbine models and size. All turbines studied here were free-standing and between 6 and 25 m (mean 16 m, 75% of sites had turbines between 15 and 18 m; including hub height in the analyses presented here does not affect results or conclusions – see Supplementary Material) in hub height, and all were in rural settings but specific habitat in the surrounding area varied (see Habitat data and variable selection). The number of SWTs installed in each site varied between one and four (median 1.6) and the distance between individual turbines in a site was between 12 and 90 m (with only one site >62 m and a mean of 46.4 m). SWTs installed in individual sites were the same size and specification with the exception of one site. More than two SWTs were installed in only five sites in our sample. We therefore restricted the analyses presented here to a comparison of single and multiple turbine sites; hence, we here refer to sites with 2–4 SWTs installed as 'multiple turbine' sites throughout the remainder of this paper. Although this limits possible inference from our findings somewhat, we are confident that this reflects the current distribution of numbers of turbines per site in the UK; that is, the majority of sites have one or two SWTs installed (pers. obs.).

Bat data and transects

Bat activity data were collected between 28 May and 1 September 2013 and 7 July and 4 September 2014. This period reflects the seasonal activity of the species studied here (Swift, 2009). The time of data collection varied but started 30 min after sunset at the earliest and finished at least 60 min before sunrise in all cases.

Bat activity was measured along transects by 1–2 observers walking the length of each transect at a slow (approximately 2.5 km h^{-1}) and constant pace, using EchoMeter EM3+ bat detectors (WildLife Acoustics, Maynard, MA, USA). A target of four transects was planned for all sites, running out from the turbine (or the central point between turbines in the case of multiple turbine sites) in four directions. However, because of physical constraints (e.g. walls, impassable fences or ditches, houses or buildings), the actual number of transects per site as well as their length varied (3–4 transects, mean 3.7 per site, length: 300–500 m). All transects were placed using a combination of GIS-based planning and on-site assessment of landscape variation, so that (1) the combination of all transects within each site covered all major habitats present, and (2) the overall distance separating each transect was maximized. Transects were divided into 100 m sections running out from the turbine centre point, giving up to five distance bands running away from the turbine. See Fig. A1 for an example transect and transect section layout.

One measure of ground level wind speed was taken at the end of each section (i.e. in each distance band) on each survey visit using handheld anemometers. Minimum daily temperature measures for each survey visit were obtained from the UK MIDAS weather station data at Grangemouth (N 56° 1′ 5.15, W 3° 43′ 5.88, between 5.5 and 92.6 km, average 52.4 km, from the sites) (UK Meteorological Office, 2006).

Habitat data and variable selection

To account for expected confounding effects of habitat variation on bat activity, we collated measurements of habitat availability per transect section. To do this, we placed 50 m buffers around digital maps of each transect route, resulting in approximately 100 m × 100 m transect sections. The exact area of each section varied because of non-linear transect sections, but this was accounted for in the analysis, see Statistical analysis and Fig. A1. All habitat variables were quantified in each transect section using 1:1250 UK

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Ordnance Survey MasterMap Topography digital maps, using QGIS v. 1.8 (http://www.qgis.org/en/site/). We focused on two key measures of habitat variability in each transect section. These two variables were selected on the basis of a preliminary analysis (See Supplementary Material) of the effect of a full set of 10 habitat variables on bat activity. This was done to avoid both overparameterization of the models, as well as the inclusion of highly collinear habitat metrics. First, we used edge density (m m^{-2}) as a measure of the density of 'edge' habitat (defined as woodland and water edges, hedgerows, roads and tracks, roadsides, field boundaries and building edges), calculated as the total length of all such features divided by the area of the transect section. Although the specific effect of these pooled habitat features on bat activity may vary, all tend to be associated with high levels of bat activity relative to open and/or homogenous habitat (e.g. through their use as commuting routes) (Walsh & Harris, 1996). Second, we used the proportion of woodland calculated as the sum of all tree coverage (m^2) (coniferous, non-coniferous and unclassified trees) divided by the transect section area.

Data analysis and statistics

Bat activity: probability of a pass per hectare surveyed

Bat activity was initially quantified as the number of bat passes (defined as a sequence of at least two 'search-phase' echolocation calls separated by less than a second) per transect section. However, we chose to analyse our data as bat activity presence or absence per transect section, per survey visit, because the distribution of observed counts was highly skewed (many zeros and excessive variation), so that count-based statistical models did not provide any reasonable fit. Thus, throughout the remainder of this paper, bat activity is measured as the probability of at least one bat pass per unit area. Foraging calls ('buzzes') were recorded in some cases, but in too small numbers to be included in the analysis presented here. In addition, because the area covered by each transect section varied slightly (see Bat data and transects), we here model the probability of detecting bat activity per hectare surveyed within each section.

Statistical analysis

We used generalized linear mixed effects models (GLMMs) (Gelman & Hill, 2007) to analyse our data.

Model structure

We constructed two models for bat activity: one *Pipistrellus pipistrellus* and one for *Pipistrellus pygmaeus* activity. The structure of both models was the same. In each case, activity was modelled as the probability of a bat pass per unit area on a given survey visit as the response variable with a GLMM with binomial errors and a complementary log-log link function (*clog-log*). This link function allowed us to include transect section area as an offset in the model on the appropriate scale (Agresti, 2013). In turn, this offset accounts for slight variation in the size of each transect section (see Bat data and transects and Fig. A1), modelling probability of a bat pass per unit area. To account for the non-independence of repeated measurements from the same site and transect sections, all models included transect nested within site as a random effect. To test our two predictions (see Introduction), we included two focal fixed factors; (1) transect section (distance bands; 0–100 m, 100–200 m, 200–300 m, 300–400 m and 400–500 m from the SWTs) and (2) turbine number (single or multiple SWTs), as well as the interaction between the two. In addition to these focal factors, we included the two selected habitat variables (edge density and proportion of woodland; see Habitat data and variable selection) as continuous covariates. Moreover, because bat activity is known to vary with weather conditions (particularly temperature and wind speed), across the season and through the night, we also included covariates for the minimum temperature over 24 h (°C), wind speed (m s^{-1}), day number (1 January = 1), time to midnight (minutes, with negative and positive values indicating before and after midnight respectively) and time to midnight^2 (to account for potential non-linear effects of the latter). Finally, we included year as a factorial predictor to account for potentially systematic differences in bat activity between the 2 years of the study. Thus, accounting for any confounding effects of habitat and environmental conditions, a transect section effect would indicate a systematic difference in bat activity as a function of distance from SWTs (Prediction 1), and an interaction between this and turbine number would indicate that the strength of this effect depends on the number of SWTs installed (Prediction 2).

To avoid overparameterization of the models, interactions other than those specified above (i.e. between confounding effects) were not considered. All inputs were standardized (centred to 0 and scaled to 1 SD) to improve performance of parameter estimation and to allow direct interpretation of relative effect sizes. Marginal and conditional R^2 were presented as measures of model fit (Nakagawa & Schielzeth, 2013). We calculated and presented model predictions on the response (measured) scale. Summary statistics of unstandardized model inputs are given in Table A1.

Model predictions and interpretation

We chose to present full models only, to avoid issues with any form of model simplification (Whittingham *et al.*, 2006; Cade, 2015). We interpreted the model outputs through assessment of (1) relative effect sizes using standardized inputs, and (2) 95% quantiles of predictions of the response at each of the levels of interest (specifically, the number of turbines and distance band) at average values for the remaining coviariates. The 95% quantiles of predicted distributions were made using N = 1000 bootstrap simulations (generating new values of both the 'spherical' random effects and the independent errors in each run) of parameter estimates of each model.

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We used R version 3.1.3 (http://cran.r-project.org/) for all statistical analyses including, in particular, package *lme4* (Bates *et al.*, 2014), to fit GLMMs, and its bootMer() function to produce parametric bootstrap samples.

Results

Over the 2 years of the study, we collected bat activity data in N = 1395 transect sections, during 78 survey visits. Most sites (N = 30) were surveyed on at least two occasions (2–6 surveys per site), but four sites were only visited once. Within this sample, N = 20 were single-turbine sites, and N = 14 were multiple turbine sites (2–4 turbines). In total, we recorded N = 1867 bat passes, of which 98.4% (N = 1838) were *Pipistrellus* spp. [N = 969 (51.9%) *P. pipistrellus*, N = 714 (38.2%) *P. pygmaeus* and N = 155 (8.3%) *Pipistrellus* sp.]. The remaining 1.6% of total calls recorded were a mixture of *Nyctalus noctula*, *Plecotus auritus* Linnaeus and *Myotis* spp., but because these species combined comprised N = 29 calls, here we focus on analyses of *Pipistrellus* spp. activity. Overall, *Pipistrellus* bat activity was detected on 466 section surveys.

Parameter estimates for the models for *P. pipistrellus* and *P. pygmaeus* are presented in Table 1. In terms of confounding effects, there were strong non-zero effects of wind speed (negative), edge density (positive) on the activity of both species, as well as an effect of time to midnight (linearly negative for *P. pipistrellus* and quadratic for *P. pygmaeus*). In addition, both the proportion of woodland present and day number affected *P. pygmaeus* activity positively, and *P. pipistrellus* activity was higher in 2014 compared to 2013.

Accounting for these effects of environmental and habitat variables, we find some evidence for an effect of turbine proximity. Specifically, this effect is strongest for *P. pipistrellus* (Table 1a), for which activity in distance band 1 (0–100 m, intercept in Table 1a) is lower than the activity in distance bands 2–5 (100–500 m). Based on parameter estimates and their errors, no such effect of turbine proximity is evident in the model for *P. pygmaeus* activity (Table 1b). Moreover, again based on parameter estimates alone, there is no evidence for an interaction between distance band and number of turbines installed.

This interpretation of coefficient estimates above is largely mirrored by assessment of the predicted levels of activity, although a strong suggestion of an interaction with the number of turbines installed is evident (Fig. 1). Predicted activity is lowest for both species in distance band 1 (0–100 m). In particular, for both species, at multiple SWT sites, the predicted mean activity (measured as the probability of a bat pass per unit area) in the first distance bands (*P. pipistrellus* (Fig. 1a): 0.057; *P. pygmaeus* (Fig. 1b): 0.109) falls outside the 95% quantiles of point estimates for all other distance bands (combined 95% quantiles for bands 2–5, *P. pipistrellus*: 0.062–0.248; *P. pygmaeus*: 0.113–0.399), whereas the means of the distance bands 2–5 all fall within each other's 95% quantiles. There is no such difference for single SWT sites: the predicted means in all distance bands fall within each other's 95% quantiles, although in the case of *P. pipistrellus* activity tends to be lower in the first distance band as well (Fig. 1a, single SWT).

Table 1 Parameter estimates (*b*) and their standard errors (SE) from generalized linear mixed effects models for the activity (probability of a pass ha^{-1}) of (a) *Pipistrellus Pipistrellus* and (b) *Pipistrellus pygmaeus*

	(a) *P. pipistrellus*		(b) *P. pygmaeus*	
Fixed effects	*b*	SE	*b*	SE
(Intercept)	**−2.203**	**0.197**	**−1.864**	**0.198**
Distance band 2 (100–200 m)	**0.521**	**0.208**	0.174	0.203
Distance band 3 (200–300 m)	**0.730**	**0.207**	0.253	0.205
Distance band 4 (300–400 m)	**0.596**	**0.213**	0.232	0.210
Distance band 5 (400–500 m)	**0.489**	**0.223**	0.306	0.211
No. turbines (multiple)	−0.080	0.367	−0.263	0.363
Min 24 h temperature	−0.041	0.150	−0.099	0.162
Day number	0.124	0.203	**0.739**	**0.195**
Year (2014)	**0.826**	**0.252**	−0.602	0.301
Time to midnight	**−0.536**	**0.156**	0.294	0.175
Time to midnight^2	−0.472	0.287	**−1.208**	**0.332**
Wind speed ($m\ s^{-1}$)	**−0.549**	**0.172**	**−0.759**	**0.191**
Edge density ($m\ m^{-2}$)	**0.670**	**0.132**	**0.620**	**0.144**
Proportion woodland	−0.067	0.141	**0.493**	**0.125**
D. band 2 * no. turbines	0.260	0.428	0.693	0.412
D. band 3 * no. turbines	0.433	0.422	0.421	0.405
D. band 4 * no. turbines	−0.379	0.424	0.461	0.411
D. band 5 * no. turbines	−0.018	0.443	0.781	0.412
Random effects σ^2				
Transect/Site	0.374		0.579	
Site	0.377		0.447	
Model fit				
Marginal R^2	0.304		0.255	
Conditional R^2	0.451		0.507	

All models included site and transect within site as a nested random effect structure (variance estimates at the bottom of the table), had binomial errors and were fitted with a complimentary log-log link. All coefficients are parameter estimates based on model fits using standardized (centred and scaled to 1 SD) inputs. Instances where the point estimate ± 2* SE does not overlap zero are highlighted in bold, indicating evidence that the parameter is different from zero.

Discussion

We show that, when accounting for confounding effects of habitat and environmental conditions, activity of *P. pipistrellus* activity is lower at 0–100 m from installed SWTs, compared to 200–500 m. Although not apparent in estimated model parameters, assessment of predictions suggests that this may also be the case for *P. pygmaeus* activity, but this effect was weaker. Similarly, the distribution of model predictions suggest that the negative effect of SWT presence in the 'closest' distance band is particularly prominent in sites with multiple SWTs installed, with an apparent absence of any difference between distance bands in the activity of *P. pygmaeus* in single SWT sites.

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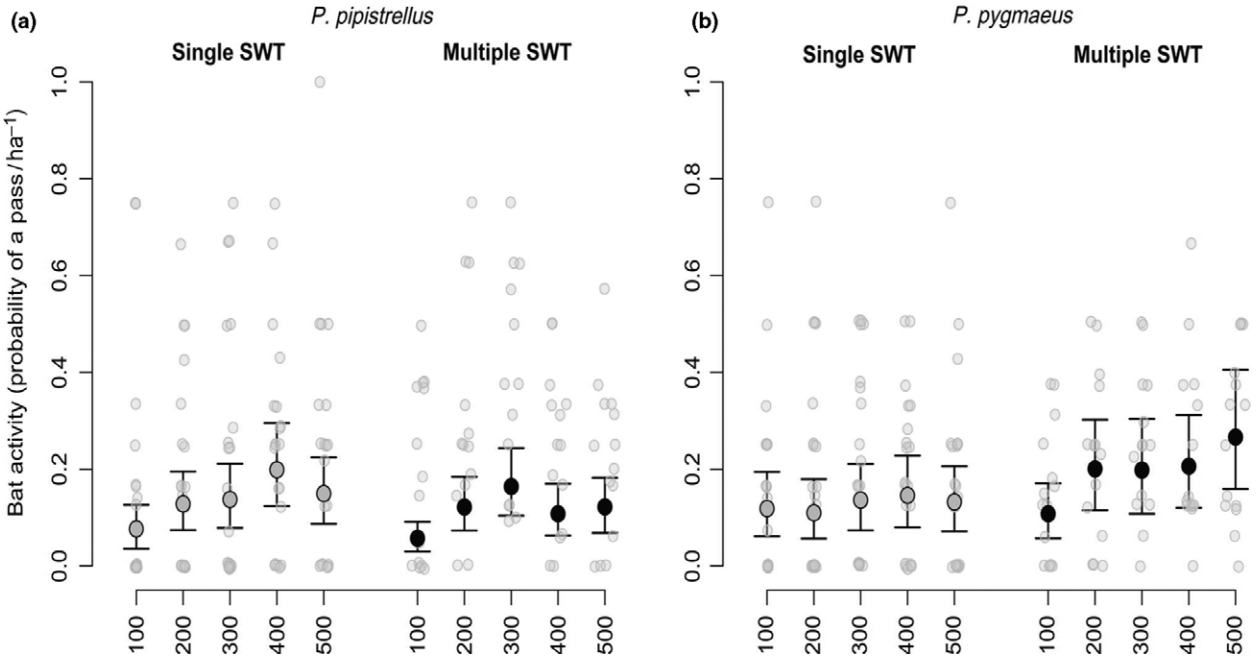


Figure 1 Observed (small grey points, average per site visit) and predicted (dark grey and black points, and lines) activity of (a) *Pipistrellus pipistrellus* and (b) *Pipistrellus pygmaeus* (probability of a pass ha^{-1}) in five distance bands from single- and multiple SWT installations. Predictions are based on model fits using standardized (scaled and centred) bat activity from generalized linear mixed effects models (GLMMs, Table 1), in each of five distance bands around small wind turbines. Dark grey circles and their corresponding lines are predicted median levels for single-turbine sites, and black circles and their corresponding lines are median levels for multiple turbine sites. Small grey dots are the observed mean probabilities of obtaining a bat pass across all observations in each distance class, for each site. The error bars (lines) are the 95% quantiles of predictions calculated from N = 1000 bootstrap samples of estimated parameter distributions.

On this basis, we conclude that the adverse effect of SWTs on *Pipistrellus* bat activity shown previously (Minderman *et al.*, 2012) may in some cases persist over longer distances in the landscape. Although here we show that differences in average predicted activity may be measured when comparing 0–100 m to 200–500 m distance bands from SWTs, it should be stressed that this does not show that adverse effects would remain evident at 100 m from SWTs. Indeed, our previous study (Minderman *et al.*, 2012) showed a potential drop in average activity of up to 50% within 25 m from operating SWTs. Even in the absence of strong effects beyond 25 m, this alone could explain the difference between 0–100 m and 200–500 m shown in this study. Nevertheless, the present findings do suggest that *Pipistrellus* bats may avoid SWTs on a wider (landscape) scale than previously demonstrated.

Although the mechanism for any avoidance behaviour is unclear, experimental studies in a laboratory setting have shown that the return of echolocation pulses off spinning turbine blades may be erratic, potentially causing navigational problems for bats (Long, Flint & Lepper, 2010). Alternatively, some bats may avoid foraging in noisy environments (Schaub, Ostwald & Siemers, 2008), but whether noise generated by turbines (including SWTs) has this effect has not been tested. In either case, it would be reasonable to assume that the strength of such interference would dissipate with distance, especially if it involves high-frequency noise (which attenuates quickly over distance), limiting the spatial scale of any adverse effect of the turbine. However, neither the extent of acoustic noise generated by SWTs nor how this operates over any distance has been studied, so it is unclear to what extent this could explain the results shown here.

Furthermore, although the model parameter estimates provide no evidence that such avoidance is stronger in sites with multiple SWTs installed for either species, model predictions of activity suggest that the decrease in activity in the 'near' distance bands tends to be stronger in multiple SWT sites, particularly for *P. pygmaeus* activity (Fig. 1b). Given known local avoidance of individual turbines (e.g. Minderman *et al.*, 2012), it is possible that clusters of SWTs installed in relatively close proximity (in our sample, between 12 and 90 m apart with mean 46.4 m and only one site >62 m and <25 m respectively) to each other have adverse effects over longer distances. For example, while local avoidance of single turbines may cause local changes of commuting routes, multiple SWTs may cause the loss of such routes altogether (similar to the effect of lighting on bat activity; Stone, Jones & Harris, 2009). However, given the mixed evidence for the interactive effect between number of turbines and proximity in this study, this suggestion needs further study. Moreover, the possibility of cumulative effects (i.e. the combined effect of multiple wind farm sites in a

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wider area, 'regionally cumulative effects') is often raised for larger turbines and wind farms (Masden *et al.*, 2010; SNH, 2012). Here, we aimed to answer a slightly different question, namely whether the installation of a larger number of SWTs installed in a single site increases the effect on bats (i.e. 'locally cumulative effects'). The findings reported here should thus not be used to draw conclusions on whether the effects of multiple SWT sites developed in a wider area could combine into a 'regional cumulative effect'. Finally, it should be stressed that in accordance with our previous findings (Minderman *et al.*, 2012), we do not find any evidence for the hypothesis that the species studied here are attracted to SWTs, as suggested for other species by Cryan *et al.* (2014). The suggestion that the interactive effect between SWT proximity and the number of SWTs installed is stronger for *P. pygmaeus* activity is in itself interesting, as it highlights the possibility that different species vary in their response to SWT presence. Although morphologically similar, the two species considered here are different in a number of key ecological aspects, including habitat preferences (Vaughan, Jones & Harris, 1997; Nicholls & Racey, 2006; Lintott *et al.*, 2015). Although our own findings reflect this (e.g. strong positive effects of proportion of woodland on *P. pygmaeus*, but not *P. pipistrellus*), whether such ecological differences between these species affect their response to SWT presence remains to be tested, and unfortunately, we did not have sufficient statistical power to do so here.

There are a number of important caveats with our findings. First of all, these data should not be generalized to species other than *Pipistrellus* spp. Although *Pipistrellus* bats are by far the commonest species in both our wider study area as well as in the specific habitats we studied (Middleton, 2006), it is likely that the collection of sufficient data to draw further species-specific conclusions would require targeted surveys. The need to collect data at a wide range of sites prevented us from doing this here. However, the effect of larger turbines in wind farms can vary between species (e.g. collision mortality can vary between species depending on environmental conditions, Baerwald & Barclay, 2011) and we therefore consider this to be a research priority for SWTs. Second, the limited number of sites with more than two turbines in our sample means that we cannot exclude the possibility that a larger number of turbines would cause stronger effects; indeed, modelling studies in birds (Schaub, 2012) suggest that this may well be the case. However, based on our own experience and anecdotal data collected during the construction of our SWT owner database (e.g. Park *et al.*, 2013), we feel that our sample of sites fairly represents the usual number of SWTs installed per site; sites with more than two and in particular more than three SWTs remain relatively rare in the UK. Finally, the combination of recording bat activity over a relatively large scale (up to 500 m from SWTs) along transects, and a distance-based approach to turbine effects, required that we analysed our data in distance bands. As noted previously, our choice of relatively 'coarse' 100 m distance bands may limit our ability to draw conclusions on the exact distance over which avoidance behaviour might operate, but small-scale effects have already been studied (Minderman *et al.*, 2012) and we here chose to focus on wider scale effects. In addition, to allow us to generalize our sampling regime across sites with single and multiple SWTs, we chose to use single starting locations for all transects. As a result, in some cases where multiple SWTs were present, this may have caused some imprecision in the allocation of individual passes to distance bands (as exact distance to individual turbines would differ by a small amount). However, our choice of transect placement minimized any such possible bias: in multiple SWT sites, transects were oriented in such a way that they were as close to perpendicular to the orientation of the SWT 'line'. In cases where this was impossible, for example, one of the transects would pass within 50 m of another turbine, the transect in question was excluded from the analyses presented here (i.e. leading to some sites with <4 transects, see Methods). Moreover, accurately allocating bat passes to even more narrow distance bands is likely to introduce false precision. Thus, collecting data on behavioural responses to turbines on a finer scale would require alternative methods.

In conclusion, we suggest that adverse effects of SWTs on *Pipistrellus* bat activity may be measurable over a greater spatial scale than previously thought (0–100 m), although combined with our previous work, it is likely that much of such avoidance behaviour occurs in relatively close proximity to turbines (<25 m). It appears that such effects are stronger when multiple SWTs are installed, particularly for *P. pygmaeus*, but this requires further study. Finally, it should be stressed that the effects reported here are on bat activity only. Although they may reflect changes in habitat use, and this may in turn lead to population-level effects, direct effects of SWT developments on the latter remain to be studied and the conclusions drawn here cannot be thus extrapolated.

The SWT sector is expanding rapidly worldwide, and as a result planners and decision makers are increasingly faced with having to advise or decide on planned developments of multiple SWTs in clusters. To date, there were no data to inform such decisions, and the findings presented here are therefore vital and timely. We reiterate our recommendation that individual SWTs within the range of sizes considered here (6–25 m hub height) should be sited away from potentially valuable bat habitat (at least >25 m, Minderman *et al.*, 2012), but based on the current findings, we further stress that particular care should be taken in landscapes where limited alternative habitats are available. This would include known roost, commuting and foraging sites. In addition, on the basis of the potential for wider landscape-scale effects on some species, we suggest that the (at least the potential for) presence of particularly vulnerable or locally rare species is assessed within 100 m from proposed sites, particularly when more than one turbine is planned.

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Appendix S1. Preliminary habitat data analysis and additional tables.

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