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

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14 ABSTRACT

15 While the effects of wind farms on bats are widely studied, effects of small wind turbines
16 (SWTs, here <25m hub height) remain understudied. SWTs are installed in a wider range of
17 habitats compared to wind farms and their effect on wildlife can therefore be different. While
18 single SWTs can adversely affect bat activity in their immediate vicinity, they are often installed
19 in clusters, and to date no data are available on whether installation of more than one turbine
20 increases the likelihood of adverse effects on bats.

21 Here, we test whether (1) SWT installations affect the activity of two species of bat (the common
22 pipistrelle *Pipistrellus pipistrellus* and soprano pipistrelle *P. pygmaeus*) on a landscape scale
23 (here defined as up to 500m from SWTs) and (2) such an effect is stronger when multiple (2-4)
24 SWTs are installed. We show that, after accounting for potentially confounding effects (e.g.
25 variation in habitat and weather), (1) mean *P. pipistrellus* activity is lower at 0-100m compared
26 to 200-500m from SWTs, and (2) the effect on *P. pygmaeus* activity tends to be similar and
27 stronger in multiple SWT sites, although evidence for the latter is limited.

28 We conclude that in some cases, adverse effects of SWTs on bat activity may be measurable
29 over longer spatial scales (within 100m) than previously thought. However, combined with
30 earlier findings, it is likely that the bulk of such effects operate within relative close proximity of
31 SWTs (<25m). Moreover, although these effects may be species-specific, with e.g. *P. pygmaeus*
32 potentially more strongly affected by multiple SWT sites, this requires further data. These
33 findings are highly relevant to decision-making aimed at minimizing any adverse effects of wind
34 turbines, specifically single- vs. multiple SWT developments, on wildlife.

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KEY WORDS: cumulative effects, microturbines, domestic turbines, renewables, renewable energy, conservation, planning

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 disproportionately greater (adverse) effects. For example, whilst 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 e.g. collision risk per turbine is relatively low (Langston & Pullan 2003, 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 25m 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 up 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). Whilst 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).

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3 59 However, these previous studies focused on a relatively limited spatial scale (e.g. within 25m of
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5 60 installed SWTs) and on single SWTs only. Crucially, it is therefore unclear whether any such
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8 61 adverse effects of SWTs on wildlife could be greater at sites with more than one turbine
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10 62 installed, i.e. whether the installation of multiple SWTs causes a disproportionately greater effect.
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13 63 This lack of understanding of the potential disproportionate effects of sites with multiple SWTs
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15 64 is particularly important for two reasons. First, the number of SWTs installed is growing rapidly
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18 65 with over 900,000 units installed worldwide by 2014 (compared to 656,000 in 2010) and this
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20 66 growth is set to continue (Pitteloud and Gsänger 2016). Secondly, SWTs are increasingly being
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22 67 installed in groups, for example between two and four turbines in close proximity. Indeed, some
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24 68 companies specifically promote the installation of multiple SWTs to maximize yield and
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26 69 eligibility for government grants or payback schemes. As a result, where SWT installation
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28 70 requires planning permission (Park, Turner & Minderman 2013), decision makers are
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30 71 increasingly asked to decide whether multiple-turbine installations would have greater effects on
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32 72 wildlife, or if limiting the number of SWTs to be installed would be an effective mitigation
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34 73 option. Currently, the evidence base for such decisions for SWTs specifically is entirely lacking.
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40 74 Here, we aim to (1) quantify the effect of SWTs on bat activity (and thus concomitant habitat
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42 75 loss) on a wider landscape scale, up to 500m from installed turbines (as opposed to 25m in
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44 76 Minderman *et al.* 2014), and (2) to test whether this effect is different between single- and
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46 77 multiple (2-4) turbine installations. Using data collected at 34 SWT sites throughout the UK we
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48 78 test the following predictions: (1) bat activity is systematically lower in closer proximity (e.g. 0-
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50 79 100m from SWTs compared to 100-200m, 200-300m, etc.) of operating SWTs, controlling for
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53 80 the effects of habitat and environmental conditions; and (2) the effect of SWT proximity on bat
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55 81 activity is stronger in sites with multiple (2-4) SWTs installed compared to single SWT sites.
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83 **METHODS**

84 **Sites**

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

102 **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 minutes after sunset at the earliest and finished at least 60 minutes 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, Mass., 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: 300-500m). 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 100m sections running out from the turbine centre point, giving up to five distance bands running away from the turbine. See Figure 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 $56^{\circ} 1' 5.15$, W $3^{\circ} 43' 5.88$, between 5.5 and 92.6 km, average 52.4km, from the sites) (UK Meteorological Office 2006).

Habitat data and variable selection

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3 125 To account for expected confounding effects of habitat variation on bat activity we collated
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5 126 measurements of habitat availability per transect section. To do this, we placed 50m buffers
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8 127 around digital maps of each transect route, resulting in approximately 100m x 100m transect
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10 128 sections. The exact area of each section varied because of non-linear transect sections, but this
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12 129 was accounted for in the analysis, see [Statistical analysis](#) and Figure A1. All habitat variables
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14 130 were quantified in each transect section using 1:1250 UK Ordnance Survey MasterMap
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16 131 Topography digital maps, using QGIS v. 1.8 (<http://www.qgis.org/en/site/>). We focused on two
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18 132 key measures of habitat variability in each transect section. These two variables were selected on
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20 133 the basis of a preliminary analysis (See Supplementary Material) of the effect of a full set of 10
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22 134 habitat variables on bat activity. This was done to avoid both overparameterisation of the models,
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24 135 as well as the inclusion of highly collinear habitat metrics. First, we used **edge density** (m m^{-2})
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26 136 as a measure of the density of "edge" habitat (defined as woodland and water edges, hedgerows,
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28 137 roads and tracks, roadsides, field boundaries and building edges), calculated as the total length of
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30 138 all such features divided by the area of the transect section. Although the specific effect of these
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32 139 pooled habitat features on bat activity may vary, all tend to be associated with high levels of bat
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34 140 activity relative to open and/or homogenous habitat (e.g. through their use as commuting routes)
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36 141 (Walsh & Harris 1996). Second, we used the **proportion of woodland** calculated as the sum of
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38 142 all tree coverage (m^2) (coniferous, non-coniferous and unclassified trees) divided by the transect
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40 143 section area.

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49 144 **Data analysis and statistics**

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52 145 ***Bat activity: probability of a pass per hectare surveyed***

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54 146 Bat activity was initially quantified as the number of bat passes (defined as a sequence of at least
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56 147 two "search-phase" echolocation calls separated by less than a second) per transect section.
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3 148 However, we chose to analyse our data as bat activity presence or absence per transect section,
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5 149 per survey visit, because the distribution of observed counts was highly skewed (many zeros and
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8 150 excessive variation) so that count-based statistical models did not provide any reasonable fit.
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10 151 Thus, throughout the remainder of this paper, bat activity is measured as the probability of at
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12 152 least one bat pass per unit area. Foraging calls (“buzzes”) were recorded in some cases, but in too
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15 153 small numbers to be included in the analysis presented here. In addition, because the area
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17 154 covered by each transect section varied slightly (see [Bat data and transects](#)), we here model the
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20 155 probability of detecting bat activity per hectare surveyed within each section.
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22 23 156 *Statistical analysis*

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25 157 We used generalized linear mixed effects models (GLMMs) (Gelman & Hill 2007) to analyse
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27 158 our data.
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30 31 159 *Model structure*

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33 160 We constructed two models for bat activity; one *P. pipistrellus* and one for *P. pygmaeus* activity.
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35 161 The structure of both models was the same. In each case, activity was modelled as the probability
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37 162 of a bat pass per unit area on a given survey visit as the response variable with a GLMM with
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39 163 binomial errors and a complementary log-log link function (*clog-log*). This link function allowed
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41 164 us to include transect section area as an offset in the model on the appropriate scale (Agresti
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43 165 2013). In turn, this offset accounts for slight variation in the size of each transect section (see [Bat](#)
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45 166 [data and transects](#) and Figure A1), modelling probability of a bat pass per unit area. To account
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47 167 for the non-independence of repeated measurements from the same site and transect sections, all
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49 168 models included transect nested within site as a random effect. To test our two predictions (see
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51 169 [Introduction](#)), we included two focal fixed factors; (1) transect section (distance bands; 0-100m,
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53 170 100-200m, 200-300m, 300-400m and 400-500m from the SWTs) and (2) turbine number (single
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3 171 or multiple SWTs), as well as the interaction between the two. In addition to these focal factors
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5 172 we included the two selected habitat variables (edge density and proportion of woodland; see
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8 173 [Habitat data and variable selection](#)) as continuous covariates. Moreover, because bat activity is
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10 174 known to vary with weather conditions (particularly temperature and wind speed), across the
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14 176 hours ($^{\circ}\text{C}$), wind speed (m s^{-1}), day number (1 January = 1), time to midnight (minutes, with
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16 177 negative and positive values indicating before and after midnight respectively) and time to
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18 178 midnight² (to account for potential non-linear effects of the latter). Finally we included year as a
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20 179 factorial predictor to account for potentially systematic differences in bat activity between the
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22 180 two years of the study. Thus, accounting for any confounding effects of habitat and
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24 181 environmental conditions, a transect section effect would indicate a systematic difference in bat
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26 182 activity as a function of distance from SWTs (Prediction 1), and an interaction between this and
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28 183 turbine number would indicate that the strength of this effect depends on the number of SWTs
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30 184 installed (Prediction 2).
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37 185 To avoid overparameterisation of the models, interactions other than those specified
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39 186 above (i.e. between confounding effects) were not considered. All inputs were standardised
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41 187 (centred to 0 and scaled to 1 SD) to improve performance of parameter estimation and to allow
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43 188 direct interpretation of relative effect sizes. Marginal and conditional R^2 were presented as
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45 189 measures of model fit (Nakagawa & Schielzeth 2013). We calculated and presented model
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47 190 predictions on the response (measured) scale. Summary statistics of unstandardised model inputs
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49 191 are given in Table A1.
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54 192 *Model predictions and interpretation*
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We chose to present full models only, to avoid issues with any form of model simplification (Cade 2015, Whittingham *et al.* 2006). We interpreted the model outputs through assessment of (1) relative effect sizes using standardised 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 covariates. 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.

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 two 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%] *Pipistrellus 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.

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3 215 Parameter estimates for the models for *P. pipistrellus* and *P. pygmaeus* are presented in Table 1.
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5 216 In terms of confounding effects, there were strong non-zero effects of wind speed (negative),
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8 217 edge density (positive) on the activity of both species, as well as an effect of time to midnight
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10 218 (linearly negative for *P. pipistrellus* and quadratic for *P. pygmaeus*). In addition, both the
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12 219 proportion of woodland present and day number affected *P. pygmaeus* activity positively, and *P.*
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14 220 *pipistrellus* activity was higher in 2014 compared to 2013.
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18 221 Accounting for these effects of environmental and habitat variables, we find some evidence for
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20 222 an effect of turbine proximity. Specifically, this effect is strongest for *P. pipistrellus* (Table 1a),
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22 223 for which activity in distance band 1 (0-100m, intercept in Table 1a) is lower than the activity in
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24 224 distance bands 2-5 (100-500m). Based on parameter estimates and their errors, no such effect of
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26 225 turbine proximity is evident in the model for *P. pygmaeus* activity (Table 1b). Moreover, again
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28 226 based on parameter estimates alone, there is no evidence for an interaction between distance
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30 227 band and number of turbines installed.
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35 228 This interpretation of coefficient estimates above is largely mirrored by assessment of the
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37 229 predicted levels of activity, although a strong suggestion of an interaction with the number of
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39 230 turbines installed is evident (Figure 1). Predicted activity is lowest for both species in distance
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41 231 band 1 (0-100m). In particular, for both species, at multiple SWT sites the predicted mean
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43 232 activity (measured as the probability of a bat pass per unit area) in the first distance bands (*P.*
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45 233 *pipistrellus* (Figure 1a): 0.057; *P. pygmaeus* (Figure 1b): 0.109) falls outside the 95% quantiles
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47 234 of point estimates for all other distance bands (combined 95% quantiles for bands 2-5, *P.*
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49 235 *pipistrellus*: 0.062-0.248; *P. pygmaeus*: 0.113-0.399), whereas the means of the distance bands 2-
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51 236 5 all fall within each other's 95% quantiles. There is no such difference for single SWT sites: the
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53 237 predicted means in all distance bands fall within each other's 95% quantiles, although in the case
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of *P. pipistrellus* activity tends to be lower in the first distance band as well (Figure 1a, single SWT).

DISCUSSION

We show that, when accounting for confounding effects of habitat and environmental conditions, activity of *P. pipistrellus* activity is lower at 0-100m from installed SWTs, compared to 200-500m. 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.

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 0-100m to 200-500m distance bands from SWTs, it should be stressed that this does not show that adverse effects would remain evident at 100m from SWTs. Indeed, our previous study (Minderman *et al.* 2012) showed a potential drop in average activity of up to 50% within 25m from operating SWTs. Even in the absence of strong effects beyond 25m, this alone could explain the difference between 0-100m and 200-500m shown in the current study. Nevertheless, the present findings do suggest that *Pipistrellus* bats may avoid SWTs on a wider (landscape) scale than previously demonstrated.

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3 260 Although the mechanism for any avoidance behaviour is unclear, experimental studies in a
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5 261 laboratory setting have shown that the return of echolocation pulses off spinning turbine blades
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8 262 may be erratic, potentially causing navigational problems for bats (Long, Flint & Lepper 2010).
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10 263 Alternatively, some bats may avoid foraging in noisy environments (Schaub, Ostwald & Siemers
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12 264 2008) but whether noise generated by turbines (including SWTs) has this effect has not been
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15 265 tested. In either case, it would be reasonable to assume that the strength of such interference
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17 266 would dissipate with distance, especially if it involves high-frequency noise (which attenuates
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19 267 quickly over distance), limiting the spatial scale of any adverse effect of the turbine. However,
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21 268 neither the extent of acoustic noise generated by SWTs nor how this operates over any distance
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23 269 has been studied, so it is unclear to what extent this could explain the results show here.
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27 270 Furthermore, although the model parameter estimates provide no evidence that such avoidance is
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29 271 stronger in sites with multiple SWTs installed for either species, model predictions of activity
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31 272 suggest that the decrease in activity in the “near” distance bands tends to be stronger in multiple
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33 273 SWT sites, particularly for *P. pygmaeus* activity (Figure 1b). Given known local avoidance of
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35 274 individual turbines (e.g. Minderman et al. 2012), it is possible that clusters of SWTs installed in
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37 275 relatively close proximity (in our sample between 12 and 90m apart with mean 46.4m and only
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39 276 one site >62m and <25m respectively) to each other have adverse effects over longer distances.
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43 277 For example, while local avoidance of single turbines may cause local changes of commuting
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45 278 routes, multiple SWTs may cause the loss of such routes altogether (similar to the effect of
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47 279 lighting on bat activity; Stone *et al.* 2009). However, given the mixed evidence for the interactive
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49 280 effect between number of turbines and proximity in the current study, this suggestion needs
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51 281 further study. Moreover, the possibility of cumulative effects (i.e. the combined effect of
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53 282 multiple wind farm sites in a wider area, “regionally cumulative effects”) is often raised for
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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 small wind turbines, 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 1997, Nicholls and 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

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3 306 between species (e.g. collision mortality can vary between species depending on environmental
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5 307 conditions, Baerwald & Barclay 2011) and we therefore consider this to be a research priority for
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8 308 SWTs. Secondly, the limited number of sites with more than two turbines in our sample means
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10 309 that we cannot exclude the possibility that a larger number of turbines would cause stronger
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12 310 effects; indeed modelling studies in birds (Schaub 2012) suggest that this may well be the case.
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15 311 However, based on our own experience and anecdotal data collected during the construction of
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17 312 our SWT owner database (e.g. Park, Turner & Minderman 2013), we feel that our sample of sites
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19 313 fairly represents the usual number of SWTs installed per site; sites with more than two and in
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21 314 particular more than three SWTs remain relatively rare in the UK. Finally, the combination of
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23 315 recording bat activity over a relatively large scale (up to 500m from SWTs) along transects, and
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25 316 a distance-based approach to turbine effects, required that we analysed our data in distance
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27 317 bands. As noted previously, our choice of relatively 'coarse' 100m distance bands may limit our
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29 318 ability to draw conclusions on the exact distance over which avoidance behaviour might operate,
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31 319 but small scale effects have already been studied (Minderman et al. 2012) and we here chose to
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33 320 focus on wider-scale effects. In addition, to allow us to generalise our sampling regime across
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35 321 sites with single and multiple SWTs, we chose to use single starting locations for all transects.
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37 322 As a result, in some cases where multiple SWTs were present, this may have caused some
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39 323 imprecision in the allocation of individual passes to distance bands (as exact distance to
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41 324 individual turbines would differ by a small amount). However, our choice of transect placement
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43 325 minimised any such possible bias: in multiple SWT sites, transects were oriented in such a way
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45 326 that they were as close to perpendicular to the orientation of the SWT "line". In cases where this
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47 327 was impossible, e.g. one of the transects would pass within 50m of another turbine, the transect
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49 328 in question was excluded from the analyses presented here (i.e. leading to some sites with <4
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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 (0-100m), although combined with our previous work it is likely that much of such avoidance behaviour occurs in relatively close proximity to turbines (<25m). 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 (6-25m hub height) should be sited away from potentially valuable bat habitat (at least >25m, 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

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species is assessed within 100m from proposed sites, particularly when more than one turbine is planned.

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Review Copy

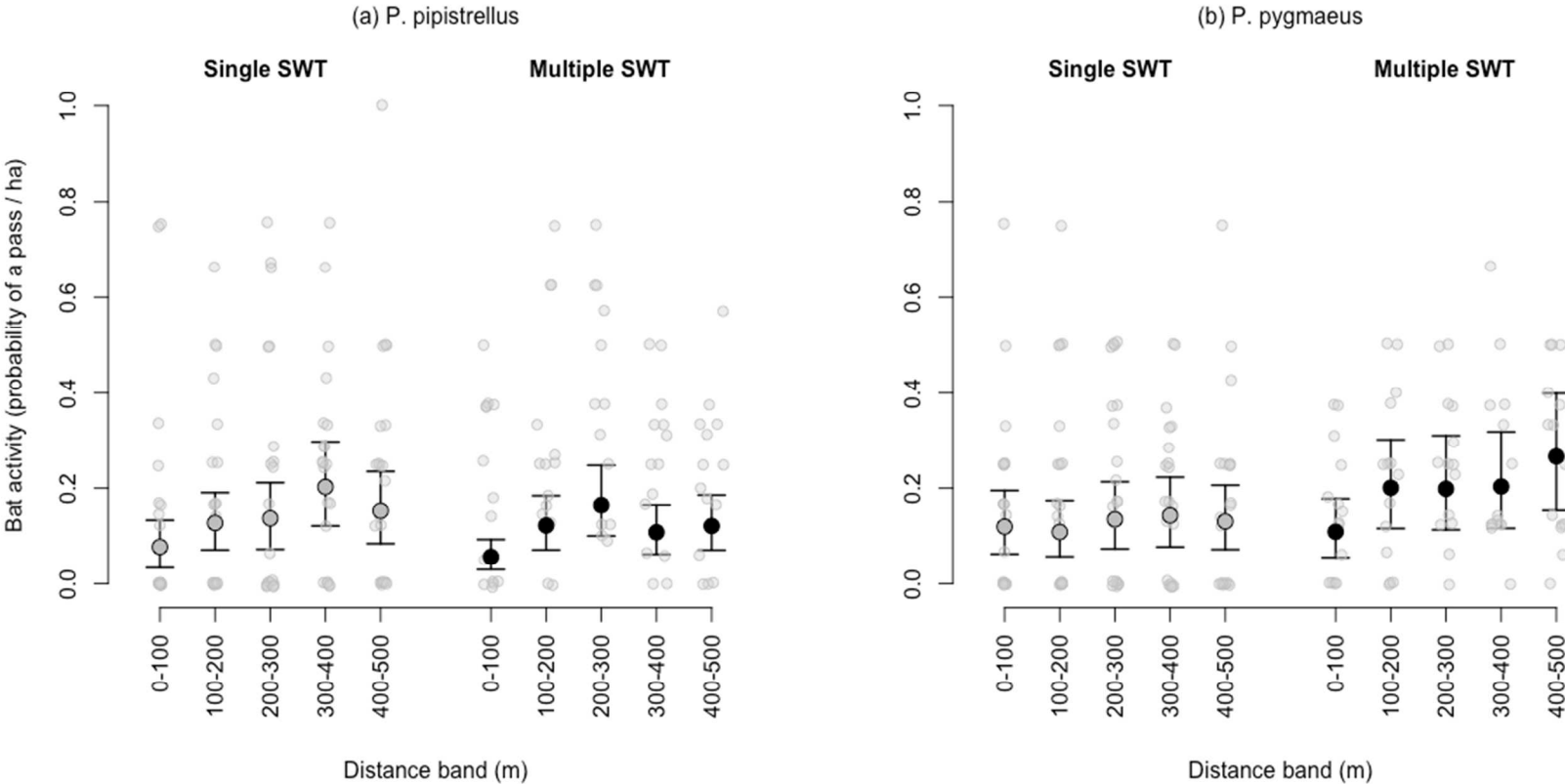
Table 1. Parameter estimates (*b*) and their standard errors (S.E.) from Generalized Linear Mixed Effects Models (GLMMs) for the activity (probability of a pass ha⁻¹) of **(a)** *P. Pipistrellus* and **(b)** *P. pygmaeus*. All models included site and transect within site as a nested random effect structure (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 standardised (centered and scaled to 1 SD) inputs. Instances where the point estimate $\pm 2 \times \text{S.E.}$ does not overlap zero are highlighted in bold, indicating evidence that the parameter is different from zero.

	(a) <i>P. pipistrellus</i>		(b) <i>P. pygmaeus</i>	
<i>Fixed effects</i>	<i>b</i>	S.E.	<i>b</i>	S.E.
(Intercept)	-2.203	0.197	-1.864	0.198
Distance band 2 (100-200m)	0.521	0.208	0.174	0.203
Distance band 3 (200-300m)	0.730	0.207	0.253	0.205
Distance band 4 (300-400m)	0.596	0.213	0.232	0.210
Distance band 5 (400-500m)	0.489	0.223	0.306	0.211
No. turbines (multiple)	-0.080	0.367	-0.263	0.363
Min 24h. 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
<i>Random effects</i> σ^2				
Transect / Site	0.374		0.579	
Site	0.377		0.447	
<i>Model fit</i>				
Marginal R^2	0.304		0.255	
Conditional R^2	0.451		0.507	

FIGURE CAPTIONS

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Figure 1. Observed (small grey points, average per site visit) and predicted (dark grey and black points, and lines) of **(a)** *P. pipistrellus*, and **(b)** *P. pygmaeus* activity (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 centered) 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.



456
457 **Figure 1.**

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

2 *Jeroen Minderman, Mairi H. Gillis, Helen F. Daly & Kirsty J. Park*

3

4 Supplementary material

5

6 Preliminary habitat data analysis and additional tables

7 A total of 10 habitat variables were quantified in each transect section in all sites (Table A2),
 8 using 1:1250 UK Ordnance Survey MasterMap Topography digital map (OS) data and QGIS v.
 9 1.8. These variables were chosen on the basis of their known potential effect on bat activity. The
 10 five proportion land cover variables (proportion buildings, road or tracks, rough grassland,
 11 roadside, and woodland) were calculated from the OS polygon area data and were calculated as
 12 the sum of each of these five land cover types in each transect section, divided by the size of the
 13 transect section. These were defined as the total area size of all polygons listed as (1) "Buildings"
 14 in the *Theme* field; (2) "Road Tracks and Paths" in the *Theme* field; (3) "Rough grassland"
 15 (including all lower classifications) in the *descrTerm* field; (4) "Roadside" in the *descrGroup*
 16 field and (5) "Coniferous or Nonconiferous trees" in the *descrGroup* field. The four distance
 17 variables (distance to buildings, linear features, trees and water) were calculated by overlaying
 18 the OS data with a 1x1m 'raster' of grid cells, calculating the distance (m) between each raster
 19 cell and the nearest raster cell containing each of the four land cover types (respectively,
 20 buildings as defined above, any linear feature as contained in the OS "line" data, any woodland
 21 as defined above and any water polygon), and averaging these distances for all cells within the
 22 transect sections.

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3 23 Finally, edge density (m m^{-2}) was expressed as the total length of all line features contained in
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5 24 the OS Linear data per transect section, divided by the transect section area. Line features
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8 25 include e.g. building-, woodland and water edges, hedgerows, roads and tracks, roadsides, and
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10 26 field boundaries.
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13 27 To identify a subset of habitat variables that were most descriptive of bat activity (see main text),
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16 28 we ran an initial model selection procedure on a Generalized Linear Mixed Effects (GLMM)
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18 29 model for the probability of a bat pass per hectare as a function of all 10 habitat variables only.
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20 30 In addition to these habitat variables this model included transect within site as a random effect,
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23 31 a binomial error distribution with a log-log link function and transect section size (ha) as an
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25 32 offset. To avoid problems with collinearity, we limited the full model set to include only one of
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28 33 the five 'proportion' habitat variables (because these represent proportion cover, lower cover of
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30 34 one automatically implies greater cover of another, leading to high collinearity). We further
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32 35 excluded all models including (1) **both** proportion of woodland and distance to trees, (2) **both**
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34 36 proportion buildings and distance to buildings, (3) **both** edge density and distance to linear
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36 37 features **or** proportion of roads or tracks. Again, these latter exclusions were to avoid high
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38 38 collinearity. For example, a greater proportion of woodland in a given transect section naturally
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40 39 implies shorter distances to trees; edge density includes linear features such roads and tracks; etc.
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42 40 Thus, these variables effectively measured very similar things but in slightly different ways. All
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44 41 model fitting procedures and analyses followed the methodology described in the main text.
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49 42 The resulting full model set comprised of a total of 112 models. The candidate set ($\Delta\text{AICc} < 4$)
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51 43 contained 4 models. The 'top' model retained three of the ten habitat variables: distance to water,
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53 44 edge density and the proportion of woodland (Table A3). Distance to water was only included in
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55 45 two out of the four models in the top set, however. By contrast, edge density and the proportion
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of woodland were retained in all four models in the candidate set. Thus, we chose to use edge density and the proportion of woodland as the key two habitat variables explaining bat activity. As discussed, we chose to limit the number of explanatory variables for the models for bat activity in this way to avoid problems with overparameterisation. However, we did test whether the inclusion of distance to water (because of the known importance of riparian habitats to *P. pygmaeus* in particular) made any difference to the models presented in Table 1 in the main text or whether it would change their interpretation. The estimated effect of distance to water was not significantly different from zero in either the model for *P. pipistrellus* (-0.259 +/- 0.151) or the model for *P. pygmaeus* (-0.043 +/- 0.181, although convergence of the estimates of this model was poor, presumably due to overparameterisation), neither was the interpretation or significance of any of the other parameters changed by its inclusions.

Similarly, we tested whether the inclusion of turbine hub height as an extra predictor, as well as its interaction with distance from turbines, affected our results. As per the addition of distance to water, this did lead to estimation problems for some of the models under consideration. Bearing this in mind, none of the interaction terms between hub height and distance from turbine were different from zero for either the *P. pipistrellus* or *P. pygmaeus* models, and when this interaction term was further dropped from both models, the effect of hub height was not different from zero in either model (*P. pipistrellus*: -0.119 +/- 0.231; *P. pygmaeus*: -0.124 +/- 0.268). The remaining parameter estimates and their interpretation also remain qualitatively unchanged.

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68 **Table A1.** Descriptive statistics of predictor variables in the Generalized Linear Mixed Model
69 (GLMM) for *Pipistrellus* sp. bat activity (Table 1), on their original (unstandardised) scale.

	Mean	Median	Std. dev	Min.	Max.
Minimum 24h temperature (°C)	11.76	12.00	2.75	5.00	17.60
Julian day number	196.59	195.00	28.85	147.00	246.00
Time to midnight (min)	-26.07	-41.00	82.54	-202.30	197.77
Time to midnight ²	7487.47	4669.44	8375.13	0.00	40925.29
Wind speed (m s ⁻¹)	1.07	0.40	1.46	0.00	9.80
Edge density (m m ⁻²)	0.04	0.03	0.02	0.00	0.13
Proportion woodland	0.05	0.00	0.13	0.00	0.92

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Table A2. Descriptive statistics of all ten original habitat variables as predictors for bat activity on their original (unstandardised) scale, split by single- and multiple SWT sites.

	Single turbines					Multiple turbines (2-4)				
	Mean	Median	SD	Min.	Max.	Mean	Median	SD	Min.	Max.
Distance to buildings (m)	174.48	152.02	114.83	13.65	596.93	222.76	209.89	135.88	9.2	623.77
Distance to linear features (m)	24.67	17.92	21.86	5.9	132.37	21.09	18.5	16.5	5.37	190.97
Distance to trees (m)	144.88	125.65	110.28	0.19	519.74	211.45	196.33	150.86	1.89	646.47
Distance to water (m)	218.3	174	166.43	13.2	689.27	186.47	153.42	142.33	17.88	652.74
Edge density (m m ⁻²)	0.04	0.03	0.03	0.00	0.13	0.04	0.04	0.02	0.00	0.13
Proportion of buildings	0.01	0.00	0.03	0.00	0.20	0.01	0.00	0.04	0.00	0.29
Proportion of roads or tracks	0.02	0.01	0.03	0.00	0.26	0.02	0.01	0.03	0.00	0.15
Proportion of rough grassland	0.07	0.00	0.18	0.00	1.00	0.02	0.00	0.07	0.00	0.49
Proportion of roadsides	0.02	0.00	0.04	0.00	0.26	0.02	0.00	0.03	0.00	0.14
Proportion of woodland	0.07	0.00	0.15	0.00	0.92	0.04	0.00	0.12	0.00	0.77

Table A3. Candidate set of Generalized Linear Mixed Models for bat activity (probability of bat activity $h a^{-1}$, binomial errors, complementary log-log link) within 4 AICc points of the “top” model, as a function of ten candidate habitat predictors only. Coefficients are parameter estimates based on models fit using standardized (centered and scaled to 1 SD) inputs. K is the number of parameters in the models. All models included site and transect within site as a nested random effect structure.

Model	Intercept	Distance to buildings	Distance to water	Edge density	Proportion woodland	K	Log-likelihood	AICc	Δ AICc	Model weight
1	-1.146		-0.305	0.523	0.41	6	-824.8	1661.7	0.000	0.541
2	-1.146	0.028	-0.301	0.532	0.408	7	-824.8	1663.7	1.962	0.203
3	-1.139			0.549	0.445	5	-826.9	1663.9	2.184	0.181
4	-1.138	0.059		0.568	0.438	6	-826.8	1665.7	3.957	0.075

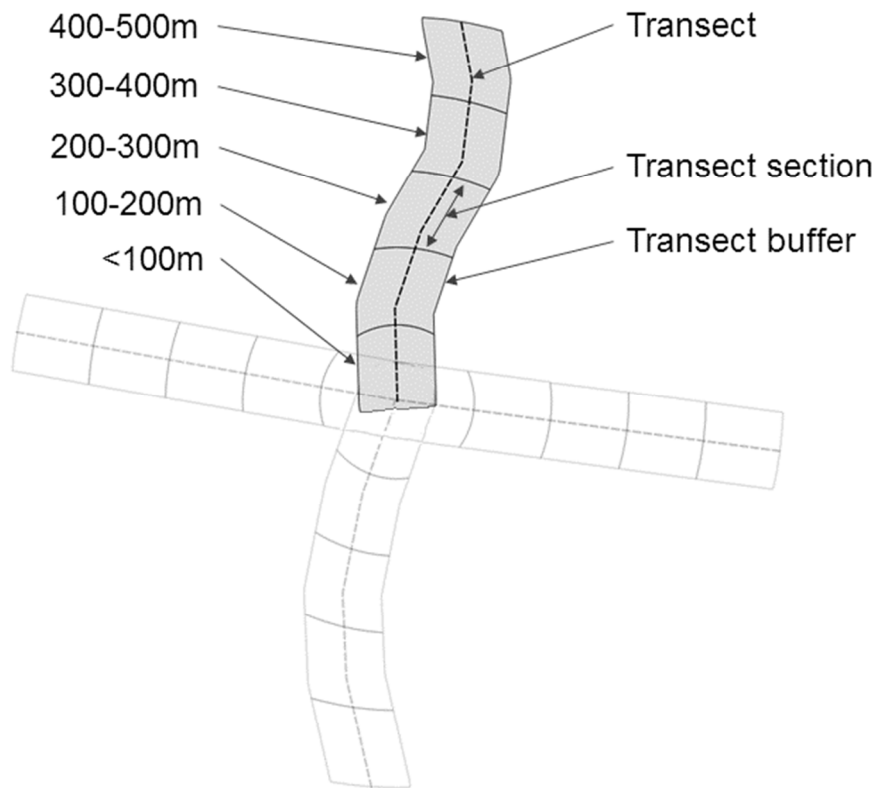


Figure A1. Conceptual example of a bat transect layout. The turbine is at the intersection of the transects. The dashed lines show the transects walked, whereas the solid line represents the area for which the habitat surrounding the turbines varied significantly between sites, and this was reflected in transect placement, so that (1) the combination of all transects within each site covered all major habitats present, and (2) overall distance separating each transect was maximized.

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