

The distribution of breeding birds around upland wind farms

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Summary

1. There is an urgent need for climate change mitigation, of which the promotion of renewable energy, such as from wind farms, is an important component. Birds are expected to be sensitive to wind farms, although effects vary between sites and species. Using data from 12 upland wind farms in the UK, we examine whether there is reduced occurrence of breeding birds close to wind farm infrastructure (turbines, access tracks and overhead transmission lines). To our knowledge, this is the first such multi-site comparison examining wind farm effects on the distribution of breeding birds.

2. Bird distribution was assessed using regular surveys during the breeding season. We took a conservative analytical approach, with bird occurrence modelled as a function of habitat, before examining the additional effects of wind farm proximity.

3. Seven of the 12 species studied exhibited significantly lower frequencies of occurrence close to the turbines, after accounting for habitat variation, with equivocal evidence of turbine avoidance in a further two. No species were more likely to occur close to the turbines. There was no evidence that raptors altered flight height close to turbines. Turbines were avoided more strongly than tracks, whilst there was no evidence for consistent avoidance of overhead transmission lines connecting sites to the national grid.

4. Levels of turbine avoidance suggest breeding bird densities may be reduced within a 500-m buffer of the turbines by 15–53%, with buzzard *Buteo buteo*, hen harrier *Circus cyaneus*, golden plover *Pluvialis apricaria*, snipe *Gallinago gallinago*, curlew *Numenius arquata* and wheatear *Oenanthe oenanthe* most affected.

5. Despite being a correlative study, with potential for Type I error, we failed to detect any systematic bias in our likelihood of detecting significant effects.

6. *Synthesis and applications.* This provides the first evidence for consistent and significant effects of wind farms on a range of upland bird species, emphasizing the need for a strategic approach to ensure such development avoids areas with high densities of potentially vulnerable species. Our results reduce the uncertainty over the magnitude of such effects, and will improve future environmental impacts assessments.

Key-words: climate change, collision, displacement, environmental impact assessment, renewable energy, upland birds

Introduction

Climate change as a result of anthropogenic climate forcing is arguably one of the greatest threats currently facing the world,

with potentially severe consequences for human society and the natural environment (IPCC 2007a, b). One of the main drivers of climate forcing is energy generation, which currently accounts for *c.* 70% of global emissions. The promotion of renewable energy sources, which contributed *c.* 15% of global primary energy supply in 2004, will be an important

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component of any emissions reduction strategy. Wind energy currently comprises *c.* 0.5% of global energy production, but there is considerable capacity to expand this to 4.8–29.1% of the energy production market by 2030 (IPCC 2007c). The proposed EU Renewable Energy Directive sets a target of 20% of energy consumption across member states to come from renewable sources by 2020 (EU Renewable Energy Directive 2008), within which the UK has been allocated a target of 15%, whilst the Scottish Government has set a target of 50% by 2020 (Scottish Government 2007). Land-based (hereafter onshore) wind farms are currently one of the cheapest and most technically developed forms of renewable energy, with considerable potential for expansion, resulting in a large number of proposed developments. The majority of these are in upland areas because these have a high wind resource, and are remote from areas of high-density human settlement.

The UK uplands support many habitats and species of high conservation importance (Thompson *et al.* 1995), and yet there is considerable uncertainty regarding the impacts of wind farms on biodiversity. Studies in other habitats and countries have highlighted the potential sensitivity of birds to wind farms, largely through birds colliding with the turbines or displacement of birds away from the turbines (Drewitt & Langston 2006, 2008; Stewart, Pullin & Coles 2007). Some poorly sited wind farms have resulted in sufficient deaths to produce a population-level effect (Barrios & Rodríguez 2004, 2007; Everaert & Stienen 2006; Smallwood & Thelander 2007; Sterner, Orloff & Spiegel 2007; Thelander & Smallwood 2007). The displacement of birds away from turbines can result in individuals abandoning otherwise suitable habitat. This has been found to occur in a number of individual wind farm studies, generally over distances of 100–200 m, although the effects vary considerably between sites and species (e.g. Leddy, Higgins & Naugle 1999; Larsen & Madsen 2000; Kowallik & Borbach-Jaene 2001; Hötter 2006; Hötter, Thomsen & Jeromin 2006; Larsen & Guillemette 2007; Devereux, Denny & Whittingham 2008).

Wind farms in the UK have not yet been associated with population-level declines because most are located away from areas with high levels of bird activity (Fielding, Whitfield & McLeod 2006; Bright *et al.* 2008). However, with more wind

farms being proposed in remote upland areas, there is growing potential for conflict between wind farm development and bird conservation, requiring a strategic approach to target such development away from concentrations of sensitive species (Bright *et al.* 2008). This requires a good knowledge of the effects of wind farms on the bird species involved, which in large part does not yet exist for most bird communities, including that of UK upland birds. Here, we model associations between wind farm infrastructure and the distribution of a range of widely distributed upland bird species across 12 wind farms. To our knowledge, this is the first attempt to identify generic patterns of the responses by breeding birds to wind farms, using a standardized method of data collection for a wide suite of species across multiple sites. As a result, our conclusions are likely to have the widest possible application to new developments, not just within the UK, but also to other semi-natural habitats with a high wind resource, such as the peatlands of north-west Europe and southern South America, and the grasslands of central North America (Archer & Jacobson 2005).

Materials and methods

SITE SELECTION

Nine wind farms located within unenclosed upland habitats (moorland, rough grassland or blanket bog) were surveyed in 2006, and a further three in 2007. We selected large (> 10 turbines) wind farms in order to maximize our ability to detect meaningful effects, encompassing a range of turbine sizes, outputs and commissioning dates (Table 1). This included most of the existing large, upland wind farms in Scotland and northern England. At each wind farm, the core survey area extended to a maximum of 1 km from the turbines, excluding areas of enclosed grassland, forest and felled forest. Because wind turbines are likely to be placed non-randomly in relation to topography, and therefore habitat, we also surveyed a 'control' area for each wind farm, located within open ground near to the turbines (range 1.3–11.4 km distance). These were selected to be as similar as possible to the habitat of the immediate turbine footprint using digital terrain data and satellite images (Buchanan *et al.* 2005). Because of time and staffing constraints, non-wind farm sites (range 180–268 ha) were smaller than wind farms (range 432–932 ha).

Site	Year of completion	Turbine hub height (m)	Number of turbines	Site capacity (MW)	Survey area (km ²)
1	2002	60	24	31.2	4.64
2	1999	35	14	8.4	4.76
3	2005	70	42	97.0	6.48
4	2006	60	17	30.0	9.32
5	1992	30	24	9.6	5.84
6	2004	60	21	48.3	7.12
7	2005	60	22	50.6	4.32
8	2000	40	26	17.2	6.24
9	1995	35	26	15.6	4.48
10	2000	40	20	13.0	8.20
11	2006	60	28	55.2	7.92
12	1996	30	36	21.6	4.72

Table 1. Summary characteristics of the 12 wind farm sites where field survey data were collected for this study

BIRD SURVEYS

Six survey visits were made at each wind farm at *c.* 12-day intervals from mid-April to the end of June, although the early visit was omitted due to access restrictions at three wind farms surveyed in 2006. Three visits were made to control sites, the date of which matched alternate surveys of the wind farms. The conditions in which surveys were conducted matched standard protocols (Brown & Shepherd 1993), being more than 3 h from dawn or dusk, and avoiding periods of strong winds, heavy precipitation and poor visibility. Observers walked transects 200 m apart across each site and plotted the locations of all moorland breeding birds on 1:12 500 maps (Pearce-Higgins & Grant 2006), excepting two abundant species; meadow pipits *Anthus pratensis* L. and skylarks *Alauda arvensis* L. for which a bespoke method was used (see the following). Sightings that were of sufficient accuracy to indicate distribution (i.e. birds first observed on the ground prior to disturbance) were separated from those of flying birds or individuals likely to have moved prior to observation, based on whether they were heard calling from a separate location to where they were first seen.

Raptor flights were mapped during these surveys to provide an index of flight activity. Once sighted, individuals were followed for a maximum of 1 min and then the same individual was not followed again until 5 min had elapsed. Thus, surveyors balanced the simultaneous demands of the breeding bird surveys and raptor flight mapping. Flight height above the ground was estimated when each bird was first seen, and changes in flight height marked on the map. To crudely assess the accuracy of plotted flight locations, field workers simultaneously estimated the location and height of eight raptor flights seen during training. Estimates of the location first seen varied by an average of $177 \pm \text{SE } 23$ m from the mean between seven observers (a surrogate for the true location), and height estimates by $20 \pm \text{SE } 4$ m from the mean. Mean height error was strongly correlated with height ($r = 0.83$, $P = 0.01$, error = $3.17 + 0.33 \times \text{height}$) indicating that it was easiest to estimate the height of low flying birds. At the start and end of each survey, field workers recorded wind direction, and estimated wind strength using the Beaufort scale.

Meadow pipits and skylarks were surveyed separately in $250 \text{ m} \times 250 \text{ m}$ subplots. Four subplots were located in a line extending outwards from the interior of the turbine array, so that the two innermost subplots contained turbines with a further two located in each control site. Subplots were surveyed on each visit by walking transects at 50-m intervals, based on the known detection function of meadow pipits and skylarks (Buchanan, Pearce-Higgins & Grant 2006), and the locations of all birds mapped with a 10-m resolution on 1:6250 maps, again separated according to accuracy.

EXPLANATORY VARIABLES

In a correlative study such as this, statistical errors in estimating the effects of wind farm infrastructure on bird distribution may occur, because wind farms are not located at random in the landscape, but mainly on exposed hill-tops and ridges. The likelihood of detecting significant wind farm related effects may therefore be influenced by species' intrinsic topographic or habitat preferences (Pearce-Higgins & Grant 2006). Therefore, we measured a wide range of potentially confounding predictor variables, including detailed measures of vegetation composition and structure made in the field, topographical (slope and altitude) and uplift data derived from a Digital Terrain Model, and assessments of the proximity to woodland cover and public roads derived from GIS layers. Each area surveyed was divided into grid squares of varying sizes according to the analysis conducted,

and values of explanatory variables aggregated within each square (Appendix S1).

The locations of turbines, above-ground transmission lines and access tracks on the wind farms were obtained from energy companies, whereas the locations of tracks on the control sites were digitized from Ordnance Survey maps. Mean distances to these features for each square, along with distance to woodland and roads, were transformed (d_x) using eqn 1 to produce a decreasing rate of decline in value with increasing distance from the turbines, simulating the likely pattern of avoidance (Finney, Pearce-Higgins & Yalden 2005). The 250 denominator virtually eliminates variation beyond 1000, equivalent to the maximum distance from the centre to the edge of the wind-farm area, preventing control sites located several kilometres from the turbines, or sites without nearby woodland cover, having undue leverage. A negative correlation with d_x indicates avoidance.

$$d_x = \exp\left(0 - \left(\frac{d}{250}\right)\right) \quad \text{eqn 1}$$

STATISTICAL ANALYSIS

Generalized Linear Models were used to test for effects of wind farm infrastructure on bird distribution, whilst controlling for the effects of confounding predictors. For grouse, waders and rarer passerines, analyses were conducted at both a fine-scale (100-m grid squares) resolution up to 500 m from the turbines using data from the core survey area only, and a large-scale (200-m grid squares) resolution employing data from both the core and control parts of each wind farm. The fine-scale analysis provides greater sensitivity to calculate potential displacement distances, whereas large-scale analysis better separates habitat from turbine proximity effects, by incorporating data from the control areas more distant from the turbines. As this second analysis may be subject to the opposite bias of control areas differing systematically from core areas in ways not measured by our predictor variables, or as a result of them being smaller and surveyed on only three occasions, we use the similarity in results from both resolutions to validate our conclusions.

Meadow pipit and skylark habitat use was analysed using grid squares of 50 m within subplots (the minimum resolution of topographical and vegetation data). Due to the error associated with locating raptor flights (above), we analysed flight activity at a 500-m resolution, averaging vegetation variables across the 100- and 200-m squares as appropriate. Both of these analyses included data from the control areas.

In each model, site was included as a 12-level fixed effect, to account for intrinsic differences in bird density between wind farms. Bird occurrence (other than raptors) was measured as the proportion of survey visits to a grid square in which an individual of a particular species was recorded, allowing us to account for the fact that some wind farms received five visits, others six, and control areas, three. Models using this response variable employed a binomial error structure and logit link function. Raptor flight activity was estimated within a grid square as the number of individual flights per survey hectare, pooled over all visits. Models using this response variable employed a Poisson error structure and log link function, with the natural log of survey effort in each unit (proportion of square surveyed \times survey visits) included as an offset. As raptors may exhibit turbine avoidance by altering their flight height, we also modelled the probability of a flight being at risk of collision within each square using a binomial regression structure, defining an 'at risk flight' as being between the lower and upper heights of the turbine span at any

point within that square. Because there was no *a priori* expectation of differing probabilities of flying at collision risk height at different locations, location was included as a random effect for this analysis, which was conducted within a Generalized Linear Mixed Model (GLMM). Only data from wind farms with a particular species present were included in analyses for that species. Due to the large numbers of zero counts in the data, which tended to be under-dispersed, scale parameters were fixed at one in all analyses.

To further guard against Type I error, we employed a two-stage modelling approach to explain variation in bird distribution that could be attributed to confounding variables, before examining effects of wind farm infrastructure (Tharme *et al.* 2001; Pearce-Higgins & Grant 2006). First, we constructed models of bird distribution using only confounding predictor variables (vegetation characteristics, topography, distance to woods and roads). This reduced the risk of a Type I error caused by failing to measure a key predictor of distribution that may itself be correlated with distance to turbine infrastructure (Lawlor *et al.* 2004a, b). We balanced this against the potential for over-fitting, by incorporating only those predictor variables that were significantly ($P < 0.05$) correlated with bird distribution in univariate tests, including both linear and quadratic terms where the quadratic term was significantly correlated with habitat use. Where predictor variables were strongly correlated ($r > 0.5$), only the most significant term was incorporated into the model. This full model was then simplified by backwards deletion to a minimum adequate model (MAM). Secondly, we tested the significance of wind farm variables (turbine, track and transmission line proximity) when separately inserted into the MAM. We did not attempt to distinguish between turbine and track proximity, because they were strongly correlated ($r = 0.79$). Throughout the modelling process, we accounted for spatial autocorrelation using an autocovariate term (Appendix S2).

We therefore took a pseudo-experimental approach to analysing the effects of wind farm variables, by assessing the probability of par-

ticular hypotheses being rejected after accounting for confounding predictor variables in the stage 1 MAM. Importantly, this allows us to be conservative with respect to the probabilities of detecting significant wind farm related effects. However, we recognize that the backwards deletion approach used to generate our stage 1 model has been criticized in recent years, in favour of information theoretic approaches (Burnham & Anderson 2002; Whittingham *et al.* 2005, 2006), and have therefore checked whether this could have affected our results (Appendix S3).

We used our results to assess those features of wind farm infrastructure to which birds respond most strongly. Thus, we model the regression slopes from Table 2 as a function of infrastructure type as a three-level factor (turbine, track or transmission line) within a GLMM, with a 19-level random effect denoting each analysis. Each case was weighted by the reciprocal of the standard error of the estimate of the slope, which means that estimated regression slopes with a high degree of error (uncertainty) are given a reduced weighting in the analysis.

For species for which there was evidence of wind farm avoidance, we assessed the likely population level effects of that avoidance by modelling bird distribution across the 12 sites as a function of turbine proximity (d_x), with all other variables fixed to mean values. Predicted probabilities, with 95% confidence intervals derived from regression standard errors, were summed within a 500-m buffer around the turbines, and compared with predictions from models in which the turbines were removed ($d_x = 0$).

Results

Models were constructed for 12 species for which there were more than 50 sightings. Stage 1 models accounted for 0–21.1% of the residual deviance in species' distribution, excluding the effects of location and the autocovariate term (Table S1).

Table 2. The effects of proximity to wind farm infrastructure on bird distribution, having accounted for other confounding variables in the stage 1 model (Appendix S2)

Species and scale	Turbine			Track			Transmission		
	Estimate	χ^2	<i>P</i>	Estimate	χ^2	<i>P</i>	Estimate	χ^2	<i>P</i>
Buzzard	-1.31 ± 0.46	8.75	0.0031	-0.86 ± 0.34	6.46	0.011	-0.26 ± 0.48	0.29	0.59
Hen harrier	-1.85 ± 0.96	3.94	0.047	-0.91 ± 0.70	1.75	0.19	-11.31 ± 30.44	0.45	0.50
Kestrel	-0.46 ± 0.39	1.45	0.23	-0.60 ± 0.30	4.05	0.044	0.44 ± 0.45	0.98	0.32
Red grouse fine scale	-0.16 ± 0.36	0.20	0.65	-0.03 ± 0.31	0.01	0.92	-0.45 ± 1.26	0.13	0.72
Red grouse large scale	0.34 ± 0.22	2.32	0.13	0.45 ± 0.19	5.54	0.019	0.55 ± 0.51	1.11	0.29
Golden plover fine scale	-1.73 ± 0.84	4.39	0.036	-1.39 ± 0.70	4.11	0.043	-0.68 ± 2.76	0.07	0.79
Golden plover large scale	-1.20 ± 0.56	4.79	0.029	-1.05 ± 0.49	4.91	0.027	-0.12 ± 1.54	0.01	0.94
Lapwing fine scale	-0.89 ± 0.99	0.82	0.36	0.20 ± 0.99	0.82	0.82	-0.21 ± 0.91	0.06	0.81
Lapwing large scale	-0.21 ± 0.52	0.16	0.69	-0.05 ± 0.48	0.01	0.92	0.26 ± 0.55	0.22	0.64
Snipe fine scale	-2.31 ± 1.21	3.90	0.048	-1.41 ± 0.91	2.43	0.12	1.22 ± 0.96	1.51	0.22
Snipe large scale	-1.61 ± 0.72	5.44	0.020	-1.06 ± 0.50	4.67	0.031	-0.12 ± 0.68	0.03	0.86
Curlew fine scale	-0.31 ± 0.51	0.38	0.54	-0.47 ± 0.44	1.08	0.30	-0.42 ± 0.51	0.71	0.40
Curlew large scale	-0.68 ± 0.30	5.31	0.021	-0.35 ± 0.25	1.99	0.16	0.24 ± 0.29	0.65	0.42
Skylark	-0.71 ± 0.38	3.62	0.057	-0.53 ± 0.37	2.09	0.15	1.61 ± 0.79	3.97	0.046
Meadow pipit	-0.40 ± 0.17	5.79	0.016	-0.18 ± 0.17	1.17	0.28	0.33 ± 0.38	0.74	0.39
Wheatear fine scale	-2.78 ± 1.28	5.01	0.025	-1.54 ± 1.02	2.35	0.13	-0.50 ± 2.27	0.05	0.82
Wheatear large scale	-1.45 ± 0.68	5.13	0.024	-0.36 ± 0.43	0.51	0.48	1.04 ± 0.58	3.02	0.082
Stonechat fine scale	0.06 ± 1.05	0.00	0.95	0.65 ± 0.85	0.60	0.44	0.0063 ± 0.99	0.00	1.00
Stonechat large scale	-0.46 ± 0.56	0.68	0.41	-0.35 ± 0.42	0.72	0.40	0.41 ± 0.49	0.69	0.41

For details of the different scales of analysis see Statistical analysis. Significant ($P < 0.05$) terms are highlighted in bold.

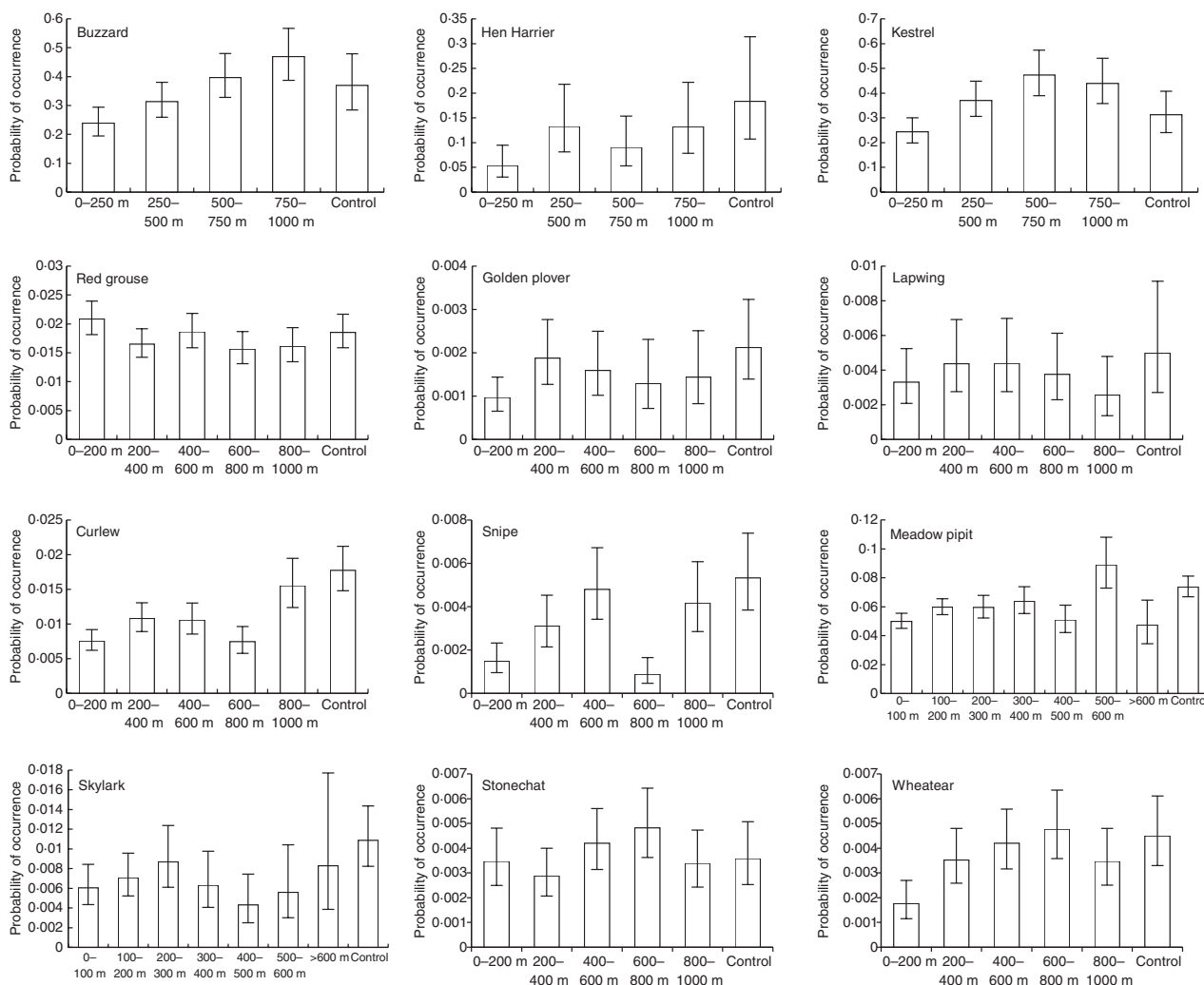


Fig. 1. Probability of species occurrence, having accounted for potentially confounding variables in the stage 1 model (Table S1). Bars represent least square means estimates (\pm SE) of the probability of a species being recorded from a grid cell on any one visit from a Generalized Linear Model with wind farm identity and distance band as factors. The resolution of the distance bands varies according to the scale of the analyses and the number of cases within each band. For example, although the analysis of meadow pipit and skylark habitat use was conducted at a 50-m resolution, data are aggregated into 100-m bands because of the limited number of cases within each 50 m band width. Conversely, the analysis of raptor habitat use was conducted at a 500-m resolution, but the data are divided into 250-m bands, based on the mean values for d_x across each square, to provide a finer degree of separation.

EFFECTS OF TURBINE PROXIMITY ON BIRD DISTRIBUTION

Turbine proximity was significantly correlated with bird distribution in 10 of the analyses (Table 2). Golden plover *Pluvialis apricaria* L., snipe *Gallinago gallinago* L. and wheatear *Oenanthe oenanthe* L. each showed significant avoidance of turbines at both fine- and large-scale resolutions, providing robust support for these relationships. Examination of the residual variation between 200-m distance bands indicate that avoidance by both golden plover and wheatear was largely restricted to 200 m, whereas avoidance by snipe extends to 400 m (Fig. 1). Additionally, there was evidence from the large-scale analysis for significant turbine avoidance by curlew *Numenius arquata* L. (Table 2), which was best explained by both a linear and quadratic term (turbine proximity slope = -2.82 ± 1.09 , $\chi^2 = 6.80$, $P = 0.009$; turbine

proximity² slope = 3.20 ± 1.55 , $\chi^2 = 4.26$, $P = 0.039$). This discrepancy between the results of the two scales of analysis for curlew resulted from the 800-m range of avoidance (Fig. 1), which extended beyond the radius covered in the fine-scale analysis. Two of the three raptors also showed significant turbine avoidance (Table 2), extending to at least 500 and 250 m from the turbines for buzzard *Buteo buteo* L. and hen harrier *Circus cyaneus* L. respectively (Fig. 1). Meadow pipits exhibited reduced occurrence within 100 m of the turbines, whereas the effects of turbine proximity on skylark distribution were of marginal significance ($P = 0.06$), extending up to 200 m (Fig. 2). When examining all possible model combinations, the top models each contained a measure of turbine proximity in the cases where our conservative analysis also highlighted a significant effect (Appendix S3). Therefore, on no occasion did our two-stage modelling using backwards deletion result in Type II error.

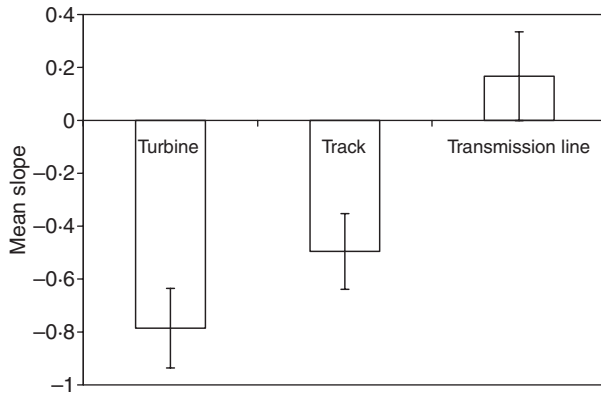


Fig. 2. Mean (\pm SE) of all regression slopes between habitat usage and proximity to turbines, tracks and transmission lines respectively.

The top models for both skylark and kestrel *Falco tinnunculus* L. also contained negative relationships with turbine proximity, indicating that these species may also avoid turbines, albeit based on this non-conservative analysis.

There were no significant effects of turbine proximity on the probability of flying 'at risk height' for any raptor species: buzzard ($F_{1,108} = 0.07$, $P = 0.79$), kestrel ($F_{1,94} = 0.14$, $P = 0.71$) and hen harrier ($F_{1,23} = 0.32$, $P = 0.75$). These null effects came after accounting for the significant effects of woodland proximity (linear: $F_{1,109} = 12.92$, $P = 0.0005$; quadratic: $F_{1,109} = 9.39$, $P = 0.0028$) for buzzard, and *Nardus stricta* L. cover (linear: $F_{1,95} = 7.28$, $P = 0.0082$; quadratic: $F_{1,95} = 7.70$, $P = 0.0067$) for kestrel.

EFFECTS OF ACCESS TRACKS AND TRANSMISSION LINES ON BIRD DISTRIBUTION

Four of the species that exhibited significant avoidance of turbines also showed significant track avoidance: buzzard, golden plover, snipe and wheatear (the latter described by a quadratic term in the fine-scale analysis; track proximity slope = -10.78 ± 4.65 , $\chi^2 = 2.35$, $P = 0.13$; track proximity² slope = 8.78 ± 4.32 , $\chi^2 = 4.26$, $P = 0.039$). Additionally, flight activity by kestrels appeared to be negatively related to track proximity, whereas the occurrence of red grouse *Lagopus lagopus scoticus* (Lath.) analysed at the large-scale was greater close to the tracks (Table 2). Significant effects of proximity to overhead transmission lines were apparent in three of the species analysed; skylark distribution varied linearly (Table 2), whereas stonechat *Saxicola torquata* L. and wheatear exhibited quadratic correlations in the large-scale analyses. Stonechat showed weak avoidance of transmission lines (linear term = -3.87 ± 1.92 , $\chi^2 = 4.33$, $P = 0.038$; quadratic term = 5.57 ± 2.36 , $\chi^2 = 5.81$, $P = 0.016$), whereas both skylark (Table 2) and wheatear (linear term = 6.56 ± 2.06 , $\chi^2 = 9.86$, $P = 0.0017$; quadratic term = -7.70 ± 2.86 , $\chi^2 = 7.59$, $P = 0.0059$) were more likely to be detected close to transmission lines.

Overall, levels of avoidance differed significantly between turbines, tracks and transmission lines ($F_{2,349} = 14.7$, $P < 0.001$). The strongest pairwise contrasts were between

turbines and transmission lines ($F_{1,345} = 28.8$, $P < 0.0001$), and tracks and transmission lines ($F_{1,352} = 15.0$, $P = 0.0005$), whereas the contrast between tracks and turbines approached significance ($F_{1,345} = 3.6$, $P = 0.066$). Thus, birds tended to avoid turbines more than tracks, but showed no evidence for the avoidance of transmission lines (Fig. 2).

EFFECTS ON DENSITY

Extrapolations of the effects of turbine proximity on the distribution of species demonstrating significant avoidance of turbines yielded predicted reductions in breeding densities of between 15% and 52%, depending upon the species (Table 3). Given the apparent avoidance distance of 800 m, we also estimated this reduction across a 1-km radius for curlew at 30.4% (3.0–52.1%).

Discussion

Our results highlight significant avoidance of otherwise apparently suitable habitat close to turbines in at least 7 of the 12 species studied, whereas there was equivocal evidence for avoidance in a further two (Appendix S3). The extent of such avoidance ranged from 100 to 800 m, but was not absolute, with modelled reductions in frequency of occurrence close to the turbines of between 20% and 80% (Fig. 3). These figures significantly reduce the current uncertainty over the magnitude of such effects.

Levels of turbine avoidance, at least for waders, were generally greater than that suggested by the few previous studies of breeding birds, but similar in magnitude to previously reported

Table 3. Predicted reductions in breeding densities, or raptor flight activity, within 500 m of the turbine array, assuming modelled habitat usage is proportional to breeding density

Species	Predicted percentage reduction in density (95% CI)
Buzzard	41.4 (16.0–57.8)
Hen harrier	52.5 (–1.2–74.2) ^a
Golden plover	38.9 (4.3–59.0)
Snipe	47.5 (8.1–67.7)
Curlew	42.4 (3.4–72.8) ^b
Meadow pipit	14.7 (2.7–25.1)
Wheatear	44.4 (4.9–65.2)

Calculations performed using the relationships in Table 2 for species exhibiting significant avoidance only.

^aThe confidence intervals for this estimate overlaps zero because despite being statistically significant based on changes in deviance, 1.96* SE of the estimate for turbine avoidance marginally overlaps zero (Table 2).

^bThe relationship between curlew habitat use and turbine proximity is best described by a combined linear and quadratic term (text). This constrains predictions of curlew habitat use to increase at distances of <200 m from the turbines (Fig. 3) but is not supported by the data (Fig. 1). We therefore capped estimates of habitat use within this distance to the minimum.

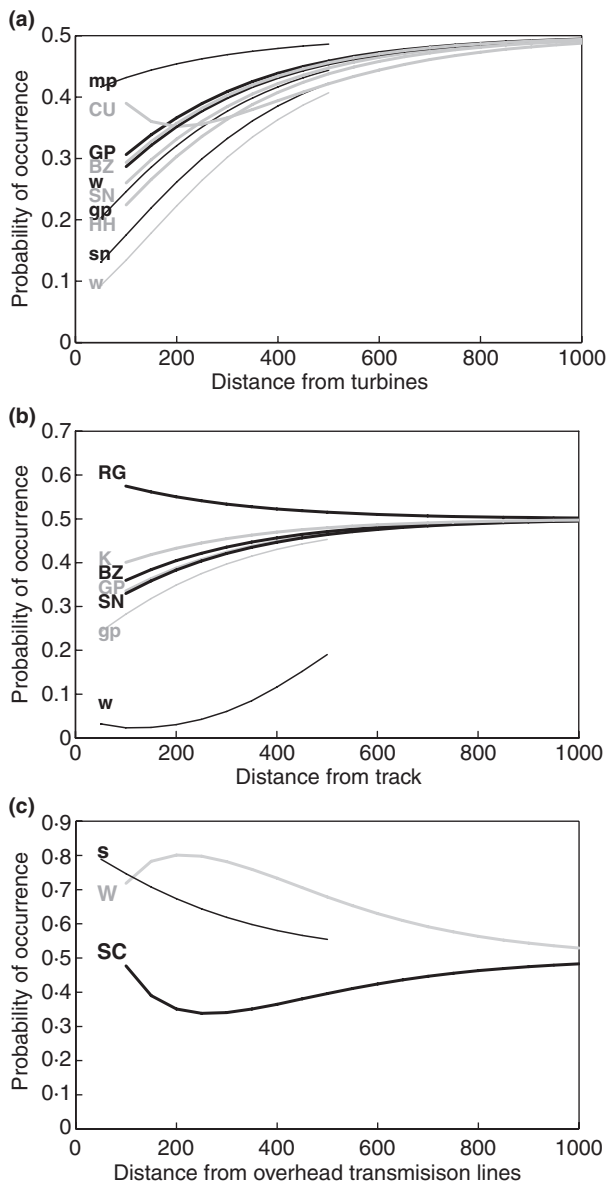


Fig. 3. Fitted significant relationships from the stage 2 analysis of species occurrence as a function of distance to (a) turbines, (b) tracks and (c) overhead transmission lines. Thin lines and lower-case codes indicate results from fine-scale and subplot analysis restricted to 50–500 m, whilst large-scale analyses restricted to 100–1000 m are denoted by thick lines and upper-case codes. Alternate black and grey lines are used to aid clarity. Intercepts are all standardized to zero (prior to logit transformation), and do not reflect differences in densities between species (bz, buzzard; hh, hen harrier; k, kestrel; rg, red grouse; gp, golden plover; cu, curlew; sn, snipe; mp, meadow pipit; s, skylark; sc, stonechat; w, wheatear).

effects on wintering habitat use (Hötter 2006; Hötter *et al.* 2006). One exception was our apparently large avoidance distance of 800 m for curlew, which suggests that breeding populations of this species may be particularly vulnerable to displacement. Previous analyses for raptors have generally found only low levels of turbine avoidance (Hötter 2006; Hötter *et al.* 2006; Madders & Whitfield 2006), with some species, such as kestrels, known to continue foraging activity close to

turbines and to be susceptible to collision (Barrios & Rodríguez 2004, 2007). We found hen harrier and buzzard showed reduced flight activity around turbines, with equivocal evidence for weak avoidance by kestrel, broadly reflecting the sensitivity of these species anticipated by Madders & Whitfield (2006). Raptors did not appear to alter their flight height in response to turbine proximity, at least at the gross scale examined. We found no evidence of turbine avoidance by red grouse. Devereux *et al.* (2008) recorded mixed effects upon wintering gamebird distribution, with pheasants *Phasianus colchicus* L., but not red-legged partridges *Alectoris rufa* L., apparently avoiding the turbines. Previous studies have detected fewer effects of turbine proximity on passerines, and where apparent, these are often smaller in magnitude, operating over 100–200 m (Leddy *et al.* 1999; Hötter *et al.* 2006; Devereux *et al.* 2008). Our findings for stonechat, meadow pipit and skylark are therefore consistent with this, although the degree of turbine avoidance by wheatear was stronger than would be inferred from other studies.

Birds showed a smaller degree of avoidance of tracks and no consistent avoidance of overhead power lines. Although we were unable to separate track from turbine avoidance in the individual models of distribution, because all turbines were located next to tracks, much of the putative avoidance of tracks may be avoidance of turbines. Although some of the species studied show avoidance of highly disturbed tracks and footpaths, such effects are only apparent at very high levels of usage (Finney *et al.* 2005; Pearce-Higgins *et al.* 2007), perhaps unlikely to be attained at many wind farms. Open country species such as skylark can show avoidance of overhead transmission lines (Milsom *et al.* 2001), whereas collision with the associated cables and towers may be a significant cause of adult mortality in other species (Bevanger 1995). Given that overhead transmission-lines were only present on 7 of the 12 wind farms, we may have had reduced power to detect such effects.

Because our study was a spatial comparison, the systematic placement of turbines with respect to habitat may result in Type I error. Reassuringly, we found no evidence for statistical bias in our analysis (Appendix S4). It is also unlikely that our results are an artefact of the planning process, which might influence energy companies to place wind farms or turbines away from areas with the greatest densities of birds. Such considerations would probably only apply to hen harrier and golden plover as the only study species listed under Annex 1 of the EU birds directive (Thompson *et al.* 1995), and therefore would not account for turbine avoidance by other species. Although individual turbines may be moved or removed from proposals if associated with a high risk to birds, such an approach is only adopted in response to collision risk modelling (Band, Madders & Whitfield 2006), and hence would probably only apply to hen harrier, as the other species would not be subject to such modelling. Alternatively, such effects might occur through the exclusion of particular sites identified as supporting high bird densities during scoping studies or initial surveys. However, this would not account for the observed fine-scale avoidance of turbines and would be expected to result in greater breeding densities at non-wind farm sites than

remote parts of the wind farms, which was not the case (Fig. 1). We can therefore have confidence in the validity of our conclusions, which being based upon a sample of 12 large wind farm sites from a wide geographical spread, are also likely to be widely applicable. However, as our analysis was a correlative, spatial comparison, it would also be valuable to compare population trends between wind farms and non-wind farm control sites to validate these conclusions.

We have not examined the mechanisms underlying our observed relationships, and we do not know whether our observations of avoidance of turbines reflect a behavioural displacement, the local population consequences of collision mortality or reduced productivity, or both. The distinction is important. If there is high mortality of birds breeding close to the turbines associated with collision, then a wind farm may become a population sink if repeatedly colonized by naïve birds. If, however, the birds simply avoid breeding close to the turbines, then depending upon the strength of density dependence (e.g. Yalden & Pearce-Higgins 1997), displaced birds may settle elsewhere with little cost or ultimately be lost to the population. Research to disentangle these mechanisms should be a high priority. However, in the absence of such mechanistic information, we suggest that our estimates of reductions in habitat usage should be used to estimate potential losses to the population and indicate likely reductions in breeding bird densities, or raptor flight activity, by up to c. 50% within 500 m of the turbines.

Conclusions

We find considerable evidence for localized reductions in breeding bird density on upland wind farms. Importantly, for the first time, we quantify the magnitude of such effects across a wide range of species, which should improve future environmental impact assessments of such development, particularly given the difficulties of collision risk modelling (Chamberlain *et al.* 2006; Madders & Whitfield 2006; de Lucas *et al.* 2008). Whether wind farms result in meaningful population-level effects at a regional and national scale depends both upon the mechanisms involved, and the overlap between bird distribution and wind farm development (Fielding *et al.* 2006; Bright *et al.* 2008; Pearce-Higgins *et al.* 2008). It is therefore important that the planning process remains robust, to ensure that wind farms, a necessary component of reducing greenhouse gas emissions, avoid sites with vulnerable bird populations (Bright *et al.* 2008), taking account of the potential for cumulative impacts (Pearce-Higgins *et al.* 2008). More widely, the contrast between our findings and those of Devereux *et al.* (2008) may suggest that species occupying remote semi-natural habitats are more sensitive to wind farm development than species occupying intensive production landscapes. In the absence of more specific information, we suggest that new wind farm developments across other similar windy semi-natural habitats in north-west Europe, southern South America, and central North America (Archer & Jacobson 2005) should therefore also avoid high densities of potentially vulnerable open country species such as waders and raptors.

Acknowledgements

This work was co-funded by RSPB, Scottish Natural Heritage and the Scottish Government, with an additional contribution from the Scottish Mountaineering Trust. We are grateful to the energy companies and land owners who granted us access to their land, and to Ewan Munro, Helen Quayle, Ellen Rotheray, Adam Seward, Sandy Watt and Ryan Wilson-Parr for their assistance with data collection. Statistical advice was provided by Stijn Bierman (Biomathematics and Statistics Scotland). Jeremy Wilson and Richard Bradbury provided many helpful comments to earlier drafts of this manuscript, and we are grateful to two anonymous referees and the editor for additional comments.

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Received 5 May 2009; accepted 23 August 2009

Handling Editor: Des Thompson

Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Explanatory variables for stage 1 minimum adequate models.

Appendix S2. Assessment of spatial autocorrelation.

Appendix S3. Information theoretic models of bird distribution.

Appendix S4. Tests of statistical bias.

Table S1. Stage 1 minimum adequate models of habitat use

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