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Poor evidence-base for assessment of windfarm impacts on birds

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SUMMARY

Concerns about anthropogenic climate change have resulted in promotion of renewable energy sources, especially wind energy. A concern raised against widespread windfarm development is that it may negatively impact bird populations as a result of bird collision with turbines, habitat loss and disturbance. Using systematic review methodology bird abundance data were synthesized from 19 globally-distributed windfarms using meta-analysis. The effects of bird taxon, turbine number, power, location, latitude, habitat type, size of area, time since operation, migratory status of the species and quality of evidence were analysed using meta-regression. Although the synthesized data suggest a significant negative impact of windfarms on bird abundance, there is considerable variation in the impact of individual windfarm sites on individual bird species, and it is unclear if the negative impact is a decline in population abundance or a decline in use owing to avoidance. Anseriformes experienced greater declines in abundance than other taxa, followed by Charadriiformes, Falconiformes and Accipitriformes, and Passeriformes. Time since windfarms commenced operation also had a significant impact on bird abundance, with longer operating times resulting in greater declines in abundance than short operating times. Other variables, including turbine number and turbine power either had very weak but statistically significant effects or did not have a significant effect on bird abundance. Windfarms may have significant biological impacts, especially over longer time scales, but the evidence-base is poor, with many studies being methodologically weak, and more long-term impact assessments are required. There is clear evidence that Anseriformes (wildfowl) and Charadriiformes (waders) experience declines in abundance, suggesting that a precautionary approach should be adopted to windfarm development near aggregations of these taxa in offshore and coastal locations. The impact of windfarm developments on bird populations must also be viewed in the context of the possible impact of climate change in the absence of windfarms.

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INTRODUCTION

The United Nations Framework Convention on Climate Change calls for stabilization of greenhouse-gas concentrations in the atmosphere to prevent dangerous anthropogenic interference with the climate (United Nations 1992). Cost-effective, carbon-emission-free technologies are required if this is to be achieved (Hoffert *et al.* 1998). Wind energy is a key element of the shift to carbon-emission-free energy, with a yearly growth rate of 30%, making it the fastest growing energy technology in the world (American Wind Energy Association 2003). However, despite the clean image of wind energy, windfarm developments may have deleterious environmental impacts (Coles & Taylor 1993; Woods 2003). In particular, attention has been brought to possible impacts on bird populations (Gill *et al.* 1996; Percival 2001; Langston & Pullan 2003; Garthe & Hüppop 2004, Barrios & Rodriguez 2004). Reported instances of habitat loss, collision mortality, displacement, disturbance and impeded movement between feeding, roosting, breeding and moulting areas all have potentially adverse impacts (Gill *et al.* 1996; Percival 2001; Langston & Pullan 2003; Garthe & Hüppop 2004; Langston *et al.* 2006), but it is unclear if these factors lead to reductions in survival or breeding productivity and ultimately to declines in the long-term abundance of bird populations. Here we formally synthesize evidence to test the hypothesis that windfarms reduce the abundance of birds in their vicinity either by displacement or population decline using a systematic review methodology, established in medicine (Khan *et al.* 2001) but rarely applied to ecological phenomena (Pullin & Knight 2001; Sutherland *et al.* 2004; Stewart *et al.* 2005) despite the widespread use of meta-analysis (Hedges & Olkin 1985; Arnqvist & Wooster 1995; Osenberg *et al.* 1999; Gurevitch & Hedges 1999, 2001; Gates 2002).

METHODS

Systematic reviews locate data from published and unpublished sources, critically appraise methodology and synthesize evidence to provide empirical answers to scientific research questions. They differ from conventional reviews in that they follow a strict methodological and statistical protocol making them more comprehensive, minimizing the chance of bias and improving transparency, repeatability and reliability (Roberts *et al*. 2006). Thus, rather than reflecting the views of authors or being based on a (possibly biased) restricted sample of literature, they provide a comprehensive assessment and summary of available evidence (Khan *et al.* 2001). Subsequent meta-analytical synthesis may increase statistical power allowing the generation of more robust and generic conclusions than those derived from single sites (Hedges & Olkin 1985; Arnqvist & Wooster 1995). Perhaps more usefully in ecology they allow exploration of potential reasons for variation in results between sites (Gurevitch & Hedges 1999) although methodological variation often confounds these relationships (Stewart *et al.* 2005; Pullin & Stewart 2006). In a systematic review context, meta-analyses also serve the useful function of defining the limits of current knowledge. Where it is difficult to derive robust guidance from meta-analytical syntheses, it is clearly inappropriate to derive generic implications from individual studies. Thus systematic reviews commonly identify knowledge-gaps highlighting the requirement for needs-led research if decision-makers are to be informed with evidence (Khan *et al.* 2001; Pullin & Stewart 2006). Guidelines for systematic review in ecology have been produced (Pullin & Stewart 2006) and further information about ecological systematic review is available at URL http://www.cebc.bham.ac.uk.

Multiple electronic databases and the internet were searched using a range of Boolean search-terms in eleven languages. The databases searched were Dogpile (URL http://www.Dogpile.com), Scirus (URL http://www.scirus.com), Copac (URL http://copac.ac.uk/) and the ISI Web of Knowledge (URL http://www.isiwebofknowledge.com/). Additional searches were performed on JSTOR the Scholarly Journal Archive (URL http://www.jstor.org/), Index to Theses Online (1970–2005; URL http://www.theses.com/) and English Nature's 'Wildlink' database (URL http://www.wildlink.org/).

Search-terms were as follows: bird* AND wind turbine*, bird* AND windfarm*, bird* AND wind park*, bird* AND wind AND turbine*, bird* AND wind AND farm*, bird* AND wind AND park*, bird* AND wind AND installation*, raptor* AND wind*, wader* AND wind*, duck* AND wind*, swan* AND wind*, geese AND wind* and goose AND wind*. Although the search-term 'wind*' encompasses the terms 'wind turbine*', 'windfarm*' and 'wind park*', initial trials proved that the number of hits become unmanageable unless the specificity of the terms was increased.

The Dogpile meta-search engine was searched using the advanced search facility, and the terms 'bird AND wind AND turbine'. It was also searched using the following languages and terms: German 'Vögel AND Windturbinen', French 'oiseaux AND turbines AND éoliennes', Spanish 'pájaros AND turbinas AND viento', Dutch 'vogels AND windturbines', Norwegian 'fugle AND vindkraft', Danish 'fugle AND vindkraft', Finnish 'lintu AND vindkraft', Swedish 'fåglar AND vindkraft', Italian 'uccelli AND vento AND turbina' and Portuguese 'pássaros AND vento AND turbina'.

These languages cover the following countries with wind energy developments: Australia, Austria, Belgium, Canada, Denmark, Finland, France, Germany, Ireland, Italy, Morocco, Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, UK, USA and others with one of these languages in official use. Internet searches are unavailable in languages of other significant wind power nations including China, Greece, India, Japan and the Ukraine, although existing English language translations from these countries were accessible. For internet searches of relevant sites, we undertook 'hand' (following links) or, where available, electronic site searches of the first 100 'hits' for each search engine within the meta-search. Articles identified by this process were assessed in the same manner as other articles.

In addition to the electronic and web searches, the library of the Royal Society for the Protection of Birds (RSPB) was hand searched. Bibliographies of articles accepted for full text viewing and relevant review articles were searched. We also contacted recognized experts and current practitioners in the fields of applied avian ecology and renewable energy technology to identify possible sources of data (including primary data) and to verify the thoroughness of our literature coverage.

Retrieved information was included providing the data pertained to bird abundance in commercial wind installations with appropriate controls or pre-development comparators. Any measure of bird abundance was considered relevant, for example bird counts whatever the spatial and temporal scale of measurement. Critical evaluation of methodology was undertaken using a hierarchy of evidence adapted from the systematic review process used in medicine and public health (Stevens & Milne 1997; Pullin & Knight 2003). Two reviewers independently assessed relevance and study quality to ensure repeatability. As well as considering the experimental design of all relevant studies, other critical data quality elements were examined in a standardized uniform manner. In particular the widespread occurrence of confounding factors resulting from variation between treatment and control at baseline or from changes concurrent with windfarm operation (ecological performance bias) were noted along with the rigour of observations as measured in terms of replication and objectivity (ecological detection bias). To test for the impact of these factors, data quality scores, summing the different aspects of data quality outlined above were added as a meta-regression co-variable (see below). Although this pragmatic approach is easy to apply, there is no measure of a study's 'true' validity (Emerson *et al.* 1990), thus caution should be exercised in interpreting the results. Full details of the data extracted, use of the data quality hierarchy and summing of data quality are provided on an article by article basis at URL http://www.cebc.bham.ac.uk/Documents/CEBC%20SR4%20Windfarm%20.pdf.

Effect sizes were calculated for individual taxa where mean, sample size and variance data regarding bird counts per unit time or bird density were available. Effect sizes were calculated using before and after monitoring data or treatment and control data post-windfarm development in the absence of robust before after control impact (BACI) data. Effect sizes are defined as the observed association between the intervention and outcome, where the change in the outcome is described as positive or negative deviation from the mean (Khan *et al.* 2001). For continuous data, effect sizes are expressed as the difference in means between treatment and control (or pre- and post-development) over pooled standard deviation (Wolf 1986; Osenberg *et al.* 1999; Khan *et al.* 2001).The pooled standard deviation is the root mean square of the treatment and control group standard deviations (Wolf 1986).

Species information within individual windfarms was combined using DerSimonian and Laird random effects meta-analyses based on weighted mean difference (WMD) (DerSimonian & Laird 1986; Cooper & Hedges 1994). The random-effects models anticipate that the taxa have genuine differences in their results and thus incorporate between-study variance in their estimates, in contrast to fixed-effects models which assume no genuine heterogeneity (Cooper & Hedges 1994). Random-effects models are more conservative than fixed-effects models, providing wider confidence intervals when there is between-study heterogeneity (Lau *et al.* 1997). Random-effects models are thus preferable, although they assume a normal distribution for the effect sizes (Deeks *et al.* 2001), which cannot be subject to transformation as the inverse variance is necessary for weighting. Species information was combined across windfarms using DerSimonian and Laird (1986) random effects meta-analyses based on standardized mean difference (SMD) with effect size estimator Hedges' adjusted *g* (Hedges & Olkin 1985). The SMD method expresses the size of the treatment effect in each trial relative to the variability observed in that trial, allowing combination of the different abundance measures used in each study, whilst Hedges' adjusted *g* provides an effect size measure with a correction factor for small sample bias (Deeks *et al.* 2001). Readers are referred to the relevant meta-analytical literature for further details (Hedges & Olkin 1985; DerSimonian & Laird 1986; Wolf 1986; Cooper & Hedges 1994; Arnqvist & Wooster 1995; Egger *et al.* 1997; Lau *et al.* 1997; Gurevitch & Hedges 1999, 2001; Osenberg *et al.* 1999; Deeks *et al.* 2001; Khan *et al.* 2001; Pullin & Stewart 2006).

The effect of including data with missing variance was explored in order to increase the comprehensiveness of the data sources and hence the applicability of the results. Missing variance was imputed using two times the largest standard deviation of existing data and average sample size resulting in conservative down-weighting (Wolf & Guevara 2001). Further sensitivity analyses were undertaken to assess the robustness of the results to aggregation bias and pseudoreplication. Windfarm grand means, sample sizes and variance were used to generate effect sizes to avoid pseudoreplication (Hurlbert 1984). Heterogeneity was assessed by formal tests of homogeneity undertaken prior to each meta-analysis (Thompson & Sharp 1999). Publication and other biases related to sample size were investigated by examination of funnel plot asymmetry (Egger *et al.* 1997). Potential sources of heterogeneity defined a priori as primary reasons for variation in effect size were (1) the species of bird, (2) the number of turbines in the installation, and (3) the power of individual turbines in the installation. The association of these factors with estimated effects were examined by performing meta-regression on data with no missing values in Stata version 8.2 (Stata Corporation, USA) using the program Metareg (Sharp 1998). Meta-regression relates the treatment effect to study-level covariates, assuming a normal distribution for the residual errors with both a within-study and an additive between-studies component of variance. Multivariate meta-regression was also performed examining the association of taxon (Accipitriformes, Anseriformes, Charadriiformes, Falconiformes and Passeriformes), turbine number and power (kW), location (inland, coastal, offshore), latitude, habitat type (marine, urban, arable, grassland, moorland, scrub or woodland/forest), size of area (km^2), time since operation (years), migratory status of the species (migrant, dispersive migrant, dispersive resident or resident) and quality of evidence (see above) with estimated effects. Unordered categorical variables were arranged in order of mean effect size for inclusion in meta-regression as an alternative to undertaking multiple *post hoc* sub-group analyses with concurrent loss of statistical power. The alpha value for interpretation of correlations between effect size and explanatory variables was arbitrarily lowered to 0.01 to control for Type 1 errors.

RESULTS

Literature searching

Systematic literature searching retrieved 2845 articles, of which 20 fulfilled relevance criteria. Five were duplicate publications based on data from the same sites, whilst 15 presented data on changes in bird abundance from independent sites and were accepted for critical appraisal. Two articles presented data on more than one windfarm, whilst one was not suitable for quantitative analysis, thus data from 19 windfarms in Europe and North America were available for synthesis. Nine of these datasets presented the means, sample sizes and variance measures necessary to calculate effect sizes, although three reported on fewer than four species. Of the remaining 10 datasets, nine did not present variance measures, one did not include windfarm characteristics and three of the sites were not independent as they shared the same control.

Meta-analysis

Random effects WMD meta-analysis of six complete independent datasets with three or more species produced negative effect sizes (i.e. reduction in local bird population density), two of which were statistically significant, although one was subject to significant bias (Table 1).

Random effects SMD meta-analysis of all nine complete datasets resulted in a pooled effect size of -0.328 ($p < 0.0001$) (Fig. 1). The inclusion of incomplete and non-independent data with down-weighted variances reduced the size of the

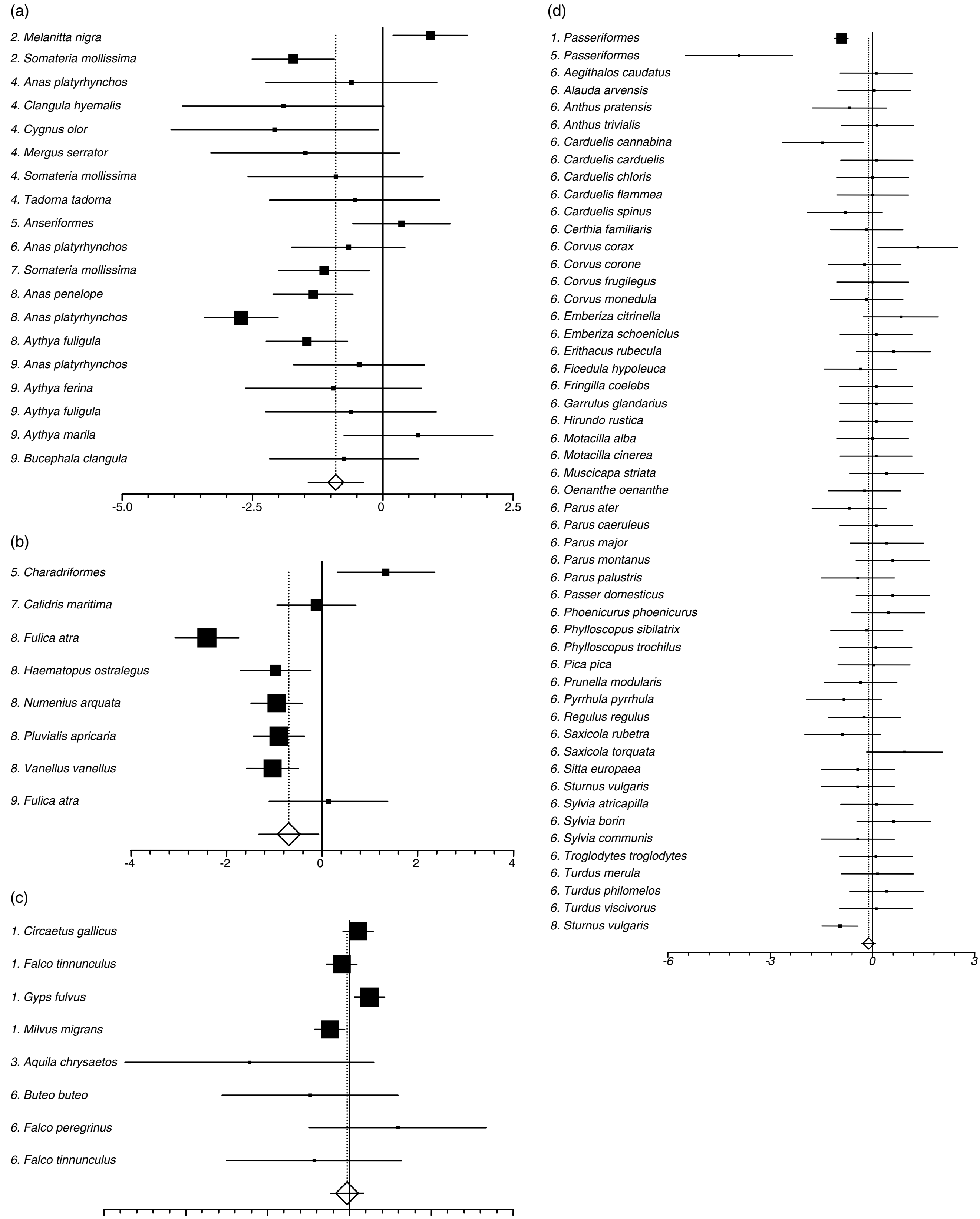

Figure 1 Forrest plot of individual species effect sizes from nine independent studies (1 = De Lucas *et al.* 2004; 2 = Guillemette *et al.* 1998; 3 = Larsson 1994; 4 = Hunt *et al.* 1995; 5 = Meek *et al.* 1993; 6 = Phillips 1994; 7 = Still *et al.* 1996; 8 = Winkelman 1992; and 9 = Winkelman 1989) with variance data pooled using standardized mean difference random effects meta-analysis. (*a*) Anseriformes, (*b*) Charadriiformes, (*c*) Accipitriformes and Falconiformes, and (*d*) Passeriformes. The x-axis is standardized mean difference. The solid vertical line represents the line of no effect; the stippled line and diamond indicate pooled effect sizes for each functional group. Box size is related to sample size, error bars are 95% confidence intervals.

Table 1 Within-study DerSimonian-Laird (DerSimonian & Laird 1986) weighted mean difference pooled effect sizes generated across species within individual studies, significance ($\alpha < 0.05$), Q statistic indicating heterogeneity and bias assessed using the Egger test. Significant results ($p < 0.01$) are in bold.

Study	*Pooled effect size*	*p*	*Q*	*Bias*	*Significant negative effect sizes*
De Lucas *et al.* (2004)	−0.699	0.383	**111.269**	−2.316	Passeriformes, *Milvus migrans*
Larsson (1994)	**−2.673**	**0.001**	8.109	1.492	*Clangula hyemalis, Mergus serrator*
Meek *et al.* (1993)	−3.762	0.762	**70.245**	insufficient strata	Passeriformes
Phillips (1994)	-5.6×10^{12}	0.999	50.918	0.046	–
Winkelman (1992)	**−275.771**	**<0.0001**	**263.339**	**−5.212**	*Anas platyrhynchos, Anas penelope, Fulica atra, Vanellus vanellus, Pluvialis apricaria, Numenius arquata, Haematopus ostralegus, Sturnus vulgaris*
Winkelman (1989)	−0.660	0.057	2.738	0.470	–

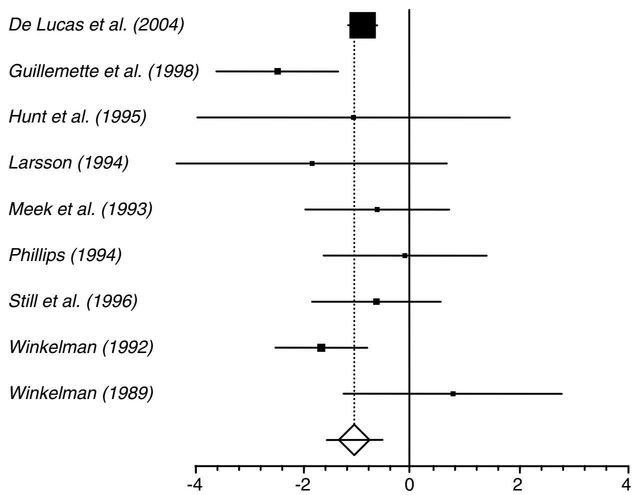

Figure 2 Forrest plot of study effect sizes based on aggregate grand means from nine independent studies with variance data, pooled using standardized mean difference random effects meta-analysis. The x-axis is standardized mean difference. The solid vertical line represents the line of no effect; the stippled line and diamond indicate pooled effect size. Box size is related to sample size, error bars are 95% confidence intervals.

effect but significance was retained (-0.033, $p = 0.002$). SMD meta-analysis of effect sizes derived from grand means of windfarms resulted in a negative and significant pooled effect-size (-0.712, $p < 0.0001$) (Fig. 2), which remained with the addition of incomplete and non-independent data with down-weighted variances (-0.257, $p = 0.023$). Although grey literature was searched, there was evidence of publication (or other) bias when all 19 datasets including imputed data were analysed resulting from inclusion of fewer small negative studies than positive ones (Egger test $= -0.303$, $p = 0.015$). However, the SMD analysis of the nine datasets with no missing data was unbiased (Egger test $= -0.297$, $p = 0.371$).

Meta-regression

There is significant heterogeneity for both WMD and SMD analyses (Table 1, SMD meta-analysis of complete data $Q = 349.958$, $p < 0.0001$). Heterogeneity can be reduced by accounting for bird taxon, which had a significant impact on the effect of windfarms on bird abundance ($r = 0.290$, $SE = 0.070$, $p = 0.0001$). Anseriformes (wildfowl) experienced greater declines in abundance than other bird groups, followed by Charadriiformes (waders), Falconiformes and Accipitriformes (raptors) and Passeriformes (songbirds). Location did not have a significant effect in multivariate meta-regression ($r = -0.494$, $SE = 0.474$, $p = 0.297$). However, bird taxon was correlated with windfarm location; sea-ducks were associated with offshore sites and Charadriiformes were often coastal (Table 2).

Wind turbine number did not affect bird abundance, whilst turbine power had a very weak, albeit significant, effect ($r = 0.002$, $SE = 0.0007$, $p = 0.004$) with low-power turbines resulting in greater declines in abundance than high-power turbines.

When all measured variables were considered, time since windfarms commenced operation had a significant impact on bird abundance ($r = 0.519$, $SE = 0.155$, $p = 0.001$) with longer operating times resulting in greater declines in abundance than short operating times.

The distance from the windfarm over which the effect is manifest is variable with sampling areas ranging from 189 km^2 to 0.5 km^2, while the majority of included studies reported only localized effects (<8 km; Table 3).

The summed data quality score was not significantly correlated with effect size.

DISCUSSION

The WMD analyses and the SMD pooled effect sizes indicate that whilst windfarms can, in some cases, have a significant and deleterious impact on local bird abundance, impacts are highly dependent on species and location. Only two windfarms (Winkelman 1992; Larsson 1994) independently showed a significant decrease in overall bird abundance, although the lack of significance on one other site (Winkelman 1989) is as likely to reflect lack of statistical power as lack of effect. Furthermore, it is unclear if the impact is a displacement effect or a decline in local population abundance. The different syntheses across windfarms produce broadly consistent results, whether the nine complete datasets or 19 datasets including imputed

Table 2 Correlation coefficients of the explanatory variables and effect size weighted by the inverse standard error of effect size. The p value is indicated in italics. Significant results ($p < 0.01$) are in bold. * = categorical variables.

Variable	*Taxon**	*Location*	*Latitude*	*Turbine number*	*Turbine power*	*Habitat type*	*Size of area*	*Time since operation*	*Migrant*	*Data quality*
Location*	**0.777**	1								
	0.00001									
Latitude	−0.152	**−0.397**	1							
	0.159	***0.0001***								
Turbine number	0.002	0.066	−0.177	1						
	0.981	*0.542*	*0.099*							
Turbine power	**0.295**	0.023	**0.809**	−0.179	1					
	0.005	*0.832*	***0.00001***	*0.096*						
Habitat type*	−0.248	0.027	**−0.726**	0.034	**−0.842**	1				
	0.02	*0.799*	***0.00001***	*0.748*	***0.00001***					
Size of area	0.119	0.15	−0.097	**0.984**	−0.034	−0.099	1			
	0.271	*0.163*	*0.367*	***0.00001***	*0.754*	*0.361*				
Time since operation	−0.255	0.034	−0.221	**0.68**	**−0.421**	0.223	**0.606**	1		
	0.016	*0.752*	*0.0391*	***0.00001***	***0.00001***	*0.037*	***0.00001***			
Migrant*	0.171	0.164	**−0.294**	0.091	−0.16	0.139	0.087	−0.002	1	
	0.113	*0.128*	***0.005***	*0.4*	*0.138*	*0.198*	*0.421*	*0.982*		
Data quality	**0.308**	0.009	**0.721**	−0.104	**0.854**	**−0.913**	0.043	**−0.424**	−0.1	1
	0.003	*0.931*	***0.00001***	*0.338*	***0.00001***	***0.00001***	*0.692*	***0.00001***	*0.353*	
Effect size	**0.371**	**0.465**	−0.138	−0.082	0.156	−0.1371	−0.02	−0.021	−0.0004	0.122
	0.0004	***0.00001***	*0.199*	*0.449*	*0.1475*	*0.2055*	*0.853*	*0.843*	*0.996*	*0.256*

data are analysed and whether combination of data involves pseudoreplication or aggregation bias. However, the inclusion of all 19 datasets resulted in funnel plot asymmetry, possibly because of publication bias. This bias arises when small negative effects of treatment are less likely to be published than large positive effects of treatment (Khan *et al.* 2001) and may reflect the problems of accessing grey literature on windfarm impacts. Client confidentiality may prevent dissemination of environmental impact assessments (EIAs) on windfarm installations, a problem experienced in the course of this review. Despite the inclusion of unpublished material, funnel plot asymmetry remains, suggesting that publication bias is accompanied by a dissemination bias. Best practice should be modified to ensure that all EIAs are disseminated, the quality of the data assessed and results incorporated with other available data. Relevant national agencies or organizations should maintain a common library of windfarm data to improve dissemination (Drewitt & Langston 2006; Langston *et al.* 2006). Publication of EIAs in the scientific literature should also be encouraged to facilitate dissemination, close scrutiny and hopefully increase standards.

Meta-regression results suggest that Anseriformes and Charadriiformes are the most vulnerable bird taxa. Anseriformes with negative effect sizes included the sea-ducks *Clangula hyemalis* (long-tailed duck), *Somateria mollissima* (eider) and *Melanitta nigra* (common scoter) (Table 1, Fig. 1); all are considered vulnerable to windfarm impacts (Gill *et al.* 1996; Langston & Pullan 2003; Garthe & Hüppop 2004), but are considered secure globally (BirdLife International 2004). However, the local reduction in abundance of these species associated with windfarms is a cause of concern given the current rate of expansion of the industry and its movement from inland to coastal and offshore sites, particularly in Europe (Garthe & Hüppop 2004).

The lack of strong significant meta-regression coefficients between effect size, turbine number and turbine power means it is not possible to resolve the debate about the relative impacts of few high-powered turbines (750 kW) versus larger numbers of smaller low-powered turbines (85 kW). Increasing turbine power may reduce the wind park area or the number of sites required, hence reducing the area subjected to potential disturbance. Turbine size has increased rapidly in recent years with turbines of 1.5–3 MW being installed routinely, at a spacing of 2–3 turbines per km^2, particularly in offshore developments (Gill *et al.* 1996; Percival 2001; Langston & Pullan 2003). The impact of these developments is an important area for further work.

The fact that longer operating times result in significantly greater declines in abundance than shorter operating times suggests that birds do not become habituated to the presence of windfarms as previously thought likely (Gill *et al.* 1996; Langston & Pullan 2003), or that local population density declines in spite of habituation. It also indicates that short-term monitoring (2–5 years) is not appropriate for the detection of declines in bird abundance. Furthermore, if this relationship persists, then windfarms could cause larger declines in bird abundance over future decades.

There are widely divergent views amongst ecologists about methods for combining the results of independent experiments (Arnqvist & Wooster 1995; Gurevitch & Hedges 1999, 2001). We chose to use a standardized mean difference effect metric commonly applied to ecological, evolutionary and medical data and used a random effects model because we were interested in examining between-study variation.

Table 3 The ecological, windfarm and methodological characteristics of the included studies. Where characteristics of independently derived effect sizes vary within studies, the reference is split numerically. Numbers following the reference denote the independent effect sizes generated within that study, for which characteristics summarized in the table vary.

Data set	*Ecological characteristics*		*Windfarm characteristics*			*Methodological characteristics*
	Location	*Taxon*	*Turbine Power (kW)*	*Turbine number*	*Time since operation (years)*	*Spatial domain of sampling and abundance measure*
De Lucas *et al.* (2004)	Southern Spain, inland	Accipitriformes, Falconiformes, Passeriformes	116	86	2	Bird counts along a 2780 m transect repeated in time in treatment and adjacent control sites.
Guillemette *et al.* (1998)	Denmark, offshore	Anseriformes	500	10		Before and after site comparison based on bird counts. The wind farm observation area is 0.8 km^2 (control 0.7 km^2).
Hunt *et al.* (1995)	California (USA), inland	Accipitriformes	85	6500	12	Site comparison based on bird counts per km^2 of road survey across an area of 189 km^2.
Johnson *et al.* (2000*a*) 1	Minnesota (USA), inland	Accipitriformes, Anseriformes, Charadriiformes, Falconiformes, Passeriformes	342	73	3	Mean abundance of birds observed during point counts averaged by 16 observations per annum over 4 years in a 13 km^2 area.
Johnson *et al.* (2000*a*) 2			750	143	2	Mean abundance of birds observed during point counts averaged by 16 observations per annum over 4 years in a 47 km^2 area.
Johnson *et al.* (2000*a*) 3				138	1	
Johnson *et al.* (2000*b*)	Wyoming (USA), inland		647	105		Bird counts replicated three times between 15 May and 31 July on 8 transects with 5 points per transect across an area of 246 km^2.
Kerlinger (2002)	Vermont (USA), inland	Accipitriformes, Falconiformes, Passeriformes	550	11		Replicated bird counts before and after windfarm construction across an area of 0.5 km^2.
Ketzenberg *et al.* (2002) 1	Saxony (Germany), coastal	Charadriiformes	550	17	4	Mean breeding pair density per 0.1 km^2 up to 1000 m from windfarm before and after installation.
Ketzenberg *et al.* (2002) 2				34		
Ketzenberg *et al.* (2002) 3				17		
Ketzenberg *et al.* (2002) 4						
Larsson (1994)	Sweden, offshore	Anseriformes	220	1	1	Mean count of bird number with before and after counts replicated on 16 and 12 occasions respectively at 3 control and 3 treatment sites within 1 km^2 of the turbine.
Meek *et al.* (1993)	Orkney (UK), inland	Anseriformes, Charadrii-formes and Passeriformes	275	2	6	Site comparison of mean number of pairs per year across an area of 0.5 km^2.
Phillips (1994)	Wales, (UK), inland	Accipitriformes, Anseriformes, Falconiformes, Passeriformes	450	22	1	Site comparison of mean no of pairs per km^2. Windfarm area 6 km^2 (control area is adjacent 8 km^2.
Schmidt *et al.* (2003)	Colorado (USA), inland				Unknown	Bird counts replicated in space and time in treatment and control sites. Mean abundance per count recorded across an unspecified area.
Still *et al.* (1996)	England (UK), coastal	Anseriformes, Charadriiformes	300	9	2	Time series with before and after commissioning mean monthly bird counts across 1 km^2.
Winkelman (1989)	Holland, coastal			25	3	Site comparison with bird counts replicated in zones within 0.5 km of windfarm site.
Winkelman (1992)	Holland, coastal	Anseriformes, Charadriiformes, Passeriformes		18	1	Site comparison of mean breeding bird number with baseline data before windfarm construction across 0.5 km^2.

Meta-regression was used to relate study characteristics to effect size. Other methods such as mixed models or fixed effect weighted regressions could have been applied and may produce variable results. However, further statistical development is required to improve statistical meta-analyses (Gurevitch & Hedges 2001) and to ascertain which methods are most applicable in which circumstances (Pullin & Stewart 2006). Pending such work, choice of meta-analytical techniques, inevitably requires considerable subjective judgment with different authors and paradigms adopting different approaches. Subjective judgment is also required in deciding when sufficient information exists for meta-analysis. Traditionally ecological meta-analysts consider the technique particularly useful where a moderate to large quantity of empirical evidence is available (Arnqvist & Wooster 1995). However, systematic reviews are guided by a priori reasoning to avoid *post hoc* rationalization and frequently highlight lack of evidence or uncertainty (Pullin & Stewart 2006). Of the five most recent medical systematic reviews published by the Cochrane collaboration (see URL http://www.cochrane.org, accessed 1 Oct 2006), all retrieved and meta-analysed fewer studies than the current review and the conclusions of four were negative, highlighting no clear evidence (Wu *et al.* 2006, Millet *et al.* 2006), considerable uncertainty (Trinh *et al.* 2006) or no significant differences (Jones *et al.* 2006). One review was akin to the current work, synthesizing few studies (six) but identifying statistically significant patterns in the data (Briel *et al.* 2006). Consideration of uncertainty has a long tradition in ecology as well as evidence-based disciplines and is implicit in the use of the precautionary principle (Underwood 1997).

Sources of uncertainty

It is a challenge to balance the uncertainty surrounding the results of meta-analysis against their potential impact when considering management implications. There are four major sources of uncertainty surrounding these results.

Firstly, the quality of the primary research hinders interpretation as there is high potential for bias. Eight datasets had potentially important confounding factors resulting from variation between treatment and control at baseline or from changes concurrent with windfarm operation. For example, rodent control was undertaken at the windfarm site in the Altamont Pass study (Hunt *et al.* 1995) but not at the control site. Additionally before and after counts were sometimes of inadequate duration to assess interannual variations in bird abundance. The most critical of these confounding effects may be the impact of food availability discussed below. Study sample sizes varied from zero to 228 replicates. Use of BACI designs was not universal, meaning effect sizes based on variation over time between treatment and control sites could not be generated thereby introducing bias associated with observational studies (site comparisons or time series). The rigour of observations was variable as measured in terms of replication and objectivity. The summed data quality score was not significantly correlated with effect size, suggesting that bifurcation of the data into high and low quality evidence was unnecessary, possibly because the low quality studies (low replication, imprecise estimates of abundance, high intra-treatment variation coupled with confounded baselines) had a high variance and therefore a low weighting in meta-analysis by inverse variance. The problems of few replicated studies, lack of comparators, inadequate duration of follow-up and poor study quality have been recognized (Langston & Pullan 2003; Drewitt & Langston 2006; Langston *et al.* 2006), and remain a hindrance in this analysis.

Secondly, the large effects on abundance within a small radius of a windfarm illustrated by the majority of included studies (Table 3) could by itself have limited biological significance, as windfarms may affect the distribution but not the population size of birds by displacing individuals to other areas without long-term negative effects on total population, assuming alternative sites are available. This is especially plausible for the areas studied in the birds' non-breeding season. However, there is concern about sea duck populations, especially around Britain (Gill *et al.* 1996; Langston & Pullan 2003; Garthe & Hüppop 2004), and even local displacement impacts could have unsustainable additive population effects on a wider scale (Langston *et al.* 2006). The cumulative impacts of windfarms could also be significant if sea ducks are displaced from the finite shallow water foraging sites suitable for windfarm construction. Therefore, although this review shows that the impacts of windfarms on birds are statistically significant, considerable uncertainty remains about whether the impacts are biologically significant and the magnitude of these impacts substantial. The small scale of the studies presents additional problems where taxa range widely and where the same areas were used to examine the effects of turbines on taxa that use space at very different scales (for example raptors and passerines), and differ in the magnitudes of their population fluctuation.

Thirdly, many of the variables investigated in the multivariate meta-regression are correlated with each other (Table 2). There are 16 statistically significant correlations between the 10 explanatory variables, some of which are categorical (Table 2), thus it is very difficult to attribute declines in bird abundance to any one variable. Both taxon and location (coastal, inland or offshore) are correlated with effect size and each other. Additionally, the existence of a relationship does not imply causality and different patterns could be generated with different variables or different classifications where categorical variables are concerned.

Fourthly, 10 windfarms were sited inland, seven were coastal and two were offshore. The robustness of conclusions regarding offshore windfarms is therefore particularly constrained by data availability (Gill *et al.* 1996; Percival 2001; Langston & Pullan 2003; Fox *et al.* 2006). The development of offshore windfarms is in its infancy and there is therefore a dearth of information in an area where it is most required. Other factors restrict the applicability of results from offshore windfarms. The flock sizes of birds in both offshore studies were small and it is believed that small flocks are less sensitive to disturbance impacts than large flocks (Langston & Pullan 2003). Additionally, the distribution of sea ducks is very variable and related to food availability (Guillemette *et al.* 1999; Percival 2001; Langston & Pullan 2003). These factors have important implications suggesting that the impact of windfarms on sea ducks may have been underestimated as the flock sizes of included studies were small, and that variability in impact may be larger than the current work predicts.

A further problem is that the original studies are concerned with testing different hypotheses. Windfarm avoidance by staging or feeding birds relates to habitat loss whereas avoidance in migrating birds (aerial habitat loss) may lower collision risk at the possible expense of increased energy expenditure. However, migratory status did not have a significant effect on the impact of windfarms on bird taxa despite variation in attachment and subsequent investment in sites (Gill *et al.* 2001).

Further work on the impact of windfarms needs to address the uncertainty arising from the sources outlined above. Meta-analytical synthesis of mortality data could determine which ecological and windfarm characteristics are associated with high mortality. It would be valuable to correlate the impact of windfarms on mortality and local population abundance to ascertain if local population declines are due to disturbance or mortality. This could be achieved by extending and updating the current systematic review and undertaking further meta-analyses. However, ascertaining the relative weights of impact due to collision mortality, habitat loss or modification and avoidance response requires considerable research effort to provide empirical data for research synthesis and to refine existing individual-based-models (West & Caldow 2006).

More primary data are required to ascertain the impact of scale effects. There is potential for long turbine strings to disrupt ecological links by displacing birds moving between feeding, breeding and roosting areas (Percival 2001; Langston & Pullan 2003; Desholm & Kahlert 2005). This could not be investigated in multivariate meta-regression as there was insufficient reporting of turbine layout for efficient and standardized data extraction and analysis. Furthermore, multiple installations may have a cumulative impact (Langston & Pullan 2003). Data from larger numbers of windfarms would be required to ascertain cumulative impacts. Recommendations regarding turbine layout and appropriate distance between individual windfarms cannot be derived from the data that are currently available.

Our analysis provides additional weight to the case being made for a better standard of EIA and post-construction monitoring of windfarm developments (Langston *et al.* 2006). Large numbers of potentially relevant studies failed to meet the inclusion criteria for the review as they did not incorporate a control or pre-development comparator. Of those with controls or pre-development comparators, nine articles, representing almost 50% of the available evidence, were unreplicated or did not report on replication. Long-term well-replicated randomized studies with established baselines and comparators are required to improve the evidence base because the scientific value of short-term unreplicated non-randomized monitoring is negligible.

Further work is also required to distinguish population change from bird displacement especially if we are to demonstrate a causal relationship between the operation of windfarms and bird abundance. This would require the development and parameterization of predictive population models.

The systematic review methodology employed here has produced a critically appraised and synthesized body of evidence indicating that windfarms sited near large aggregations of Anseriformes, and to a lesser extent Charadriiformes, risk deleterious impacts on these taxa. However, we speculate that the negative effects of windfarms on birds may be small when compared with effects of climate change. Continued reliance on fossil-fuel consumption may result in global costs to bird populations that vastly outweigh any effects of windfarms, the impact of which can be minimized by appropriate use of further research and rigorous assessment of windfarm planning applications.

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